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WATER QUALITY RESEARCH JOURNAL OF CANADA

THEME ISSUE

Innovative Approaches to Stormwater Management in Canada

Special Issue Editors

Hans Schreier
University of British Columbia
Institute for Resources & Environment

Jiri Marsalek
Environment Canada
Water Science and Technology Directorate

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The Canadian Water Network and the Canadian Mortgage and Housing Corporation supported a Canada-wide project on innovative stormwater management; the papers in this issue are a result of this initiative.

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OVERVIEW OF THE THEME ISSUE

Innovation in Stormwater Management in Canada: The Way Forward

Jiri Marsalek[1]* and Hans Schreier[2]
SPECIAL ISSUE EDITORS

[1]*Water Science and Technology Directorate, Environment Canada, Burlington, ON L7R 4A6*
[2]*Institute for Resources & Environment, University of British Columbia, Vancouver, BC V6T 1Z4*

Rapid urban expansion, increased traffic, ageing infrastructure, greater climatic variability, and the need for enhanced sustainability of urban water resources pose significant challenges to conventional stormwater management. Innovative approaches are needed in order to mitigate the risk of flooding, pollution, and aquatic ecosystem degradation, and enhance beneficial uses of urban waters. To examine such approaches, a series of three regional conferences on innovative stormwater management were held in Vancouver, Calgary, and Toronto during 2007 to 2008 under the sponsorship of the Canadian Water Network (CWN) and the Canadian Mortgage and Housing Corporation (CMHC). Authors of selected conference papers providing information on innovative approaches to mitigating the risk of flooding and reducing pollution impacts at the property, neighbourhood, and watershed scales were then invited to submit journal papers, and those accepted in the review process were included in this Special Issue of the *Water Quality Research Journal of Canada*.

An overview of the selected papers indicates that no single innovative measure is adequate under all circumstances, and a multibarrier approach is deemed to be most effective. Examples of innovations at the property level include harvesting roof runoff and reusing water, managing rainwater by infiltration in swales and into soils in bioretention areas, minimizing impervious surfaces, and using pervious pavement. At the neighbourhood level, runoff impacts are mitigated by designing roads without curbs, gutters, and drain pipes, and diverting runoff into infiltration channels, swales, and wetlands. Creating roads and parking lots with pervious pavement and draining runoff from such surfaces into infiltration basins is also discussed. Among stormwater quality source controls, potential effects of street sweeping on runoff quality enhancement were assessed. New innovations at the watershed scale include: (a) the creation of wide riparian buffer zones that can detain water, remove sediments, and mitigate nutrient export and other pollutant effects, (b) the minimization of channelization of streams and rivers, and (c) the designation of floodwater storage areas. A new water balance model that is linked to a global information system (GIS) and works at all the three scales offers the best option to conceptualize stormwater problems, and their mitigation, in urban watersheds. Finally, the aim of this Special Issue is to promote examples of successful innovative approaches to improving stormwater management in Canadian cities, hoping that other practitioners will build on this experience and bring stormwater management practice to the next higher level.

Introduction

Rapid urban expansion leads to increased traffic and greater imperviousness which result in flashier stormwater runoff and greater generation of pollution from urban nonpoint sources (Marsalek et al. 2008). There is also clear evidence that, as a result of global climate change, we are experiencing greater climatic variability (IPCC 2007). The combined effect of rapid land use changes and increased climatic variability point to the problem that conventional stormwater management systems that were designed to remove runoff via piped conveyance systems are no longer adequate to deal with larger and more intense storm events and the associated pollution. Considerable research efforts have been invested in developing alternative approaches to conventional stormwater management focusing on rainwater management, infiltration of rainfall on site, and detention of runoff during large storm events, rather than piping it directly into urban streams (U.S. EPA 2000; Stephens et al. 2002; U.S. Department of Defense 2004).

Unfortunately, many of the innovative approaches have not been sufficiently promoted and have so far not been widely incorporated in new urban developments. To remedy this situation, The Canadian Water Network (CWN-NCE) and the Canadian Mortgage and Housing Corporation (CMHC) sponsored in 2006 a two-year knowledge translation project that aimed to highlight successful innovative approaches to stormwater management across Canada, and to present case studies to practitioners and developers in Vancouver, Calgary, and Toronto during 2007 and 2008. The specific aims were to promote innovations across Canada, to identify which approaches work best under different climatic conditions, and to examine innovations in a comprehensive manner with a focus on using a multibarrier approach that is capable of reducing the anthropogenic and environmental impacts of flooding and pollution. Quite deliberately, the leaders of this project, Hans Schreier and Jiri Marsalek, decided to use the term "innovative stormwater management" rather than adopting one of the more trendy expressions like "low impact development," "water sensitive urban design," or "sustainable urban drainage systems," because they felt that the first term reflects better the historical context and evolution of managing rainwater and surface runoff in urban areas, and the newer terms are somewhat ambiguous with respect to the qualifying adjectives, "low," "water sensitive," and "sustainable."

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* Corresponding author: Jiri.Marsalek@ec.gc.ca

There has been a shift from blue water management to green water management thinking in urban watershed management (Novotny and Brown 2007). Blue water is the proportion of rainfall that runs off the land surface into lakes and rivers and recharges groundwater. In the past, the rapid removal of surface runoff has been given the majority of attention in urban stormwater management. The new approach is to focus on the green water component of the hydrological cycle, which aims at intercepting, infiltrating, detaining, and evapotranspiring as much of the rainfall as possible rather than conveying surface runoff into pipes and streams (Ellis 2008). This approach can also be described as preserving the local onsite water balance (U.S. Department of Defense 2004). This not only reduces the amount of stormwater available for runoff, but also reduces the pollution from urban nonpoint sources that enters local streams. In this new approach the focus is primarily on smaller rain events which represent, in most cases, 70 to 85% of all annual rainfall events, and possibly a slightly higher percentage of the annual runoff volume (Urbonas and Roesner 1993). Note however that in spite of this shifting emphasis, the stormwater management infrastructure still has to deliver on all design objectives, which can generally be listed as local water balance maintenance, flood protection (i.e., for large events), erosion and sediment control, stormwater quality enhancement, and beneficial use of stormwater (MOE 2003).

Innovative stormwater management and its planning should begin with rainwater management at the site or individual property level. The next consideration is to link these activities to the neighbourhood level, and then to scale them up to the watershed level.

Innovative Stormwater Management at the Property Scale

The dominant trend in property development has been to build larger houses with double garages and paved driveways, patios, and walkways on urban lots which, over the years, have been becoming smaller. This resulted in an increased extent of impervious surfaces, with less infiltration and much greater surface runoff. Runoff from such properties was traditionally drained into a dense system of urban stormwater pipes that conveyed the runoff into local drainage systems and streams. With increased urbanization and climatic variability, new approaches are needed to mitigate these problems. The innovative approaches that need to be considered at the site or property scale level are summarized in Table 1 and consist of a wide range of collection and infiltration systems that in the best cases result in zero surface runoff from the properties.

TABLE 1. Innovative approaches to rainwater management at the property level

Traditional Approach	*Innovative Approach*
1. Roof runoff is directly conveyed to storm sewers	1. Install green roofs that detain rainfall, allow some evapotranspiration, and reduce and delay storm runoff
2. Collect roof rainwater and discharge it into storm sewers	2. Connect roofwater downspouts to rainbarrels or a storage tank and use the water for indoor or outdoor uses
3. Pave driveways and walkways	3. Minimize impervious surfaces, use pervious pavement, and infiltrate runoff in swales
4. Strip topsoil, allow soil compaction during house construction and roll out thin turf layer after construction	4. Prevent soil compaction, or restore soil porosity after construction, and specify at least 30 cm of topsoil before planting lawn
5. Use piped drinking water for watering lawns and gardens	5. Use collected roof water to water lawns and gardens or develop xeriscapes
6. Remove large trees because of risk of house damage during storms	6. Plant and maintain trees on property for stormwater generation reduction and carbon credit

The first type of innovation is to collect roofwater in rainbarrels and use the collected water for gardening and lawn watering, or use the water indoors to flush toilets and wash cloth. The need for this approach, and its practical implementation, at both single-family residential and commercial scales, is described in a paper by Farahbakhsh et al. (2009) on *Developing Capacity for Large-Scale Rainwater Harvesting in Canada.*Green roofs have been heavily promoted in Europe, not only to detain and reduce runoff but also to assist in reducing energy needs within the building. The effectiveness and challenges in developing functional green roofs in different climatic conditions is discussed by Van der Linden and Stone (2009) in a paper entitled *Wet Weather Performance of a Green Roof in Waterloo, Ontario*, and by Van Seters et al. (2009) in a paper on *Green Roof Runoff Quantity, Runoff Quality, and Growing Media.*

Trees planted on individual properties can have a significant favourable impact not only on moderating air temperatures and sequestering carbon, but also on intercepting rainfall and reducing and delaying runoff. The effectiveness of trees in enhancing the local hydrological abstractions (and potentially reducing runoff) is described by Asadian and Weiler (2009) in their paper on *A New Approach in Measuring Rainfall Interception by Urban Trees in Coastal British Columbia.*

Building pervious driveways allowing rainwater or stormwater to infiltrate into the underlying soils represents another approach that can be used both at the property and neighbourhood scales. Pervious pavements serve not only to reduce surface runoff, but also to improve its quality, e.g., by removing sediment and suspended solids. Performance of an innovative permeable pavement in sediment removal is discussed in a paper by Brown et al. (2009) on *Characteristics of Sediment Removal in two Permeable Pavements.*

Another approach to managing the onsite water balance, described at the project conferences but not covered in this issue, is soil moisture retention by keeping the topsoil layer on residential lots at least 30-cm thick. This is one of the most effective measures at the property level for minimizing runoff generation, storing significant quantities of water in soils, and reducing the irrigation requirements by at least 1/3, depending on location.

Any additional stormwater that cannot be controlled by these suggested measures can be infiltrated by building rock pits (infiltration wells) on the property and connecting them with infiltrations systems (swales, raingardens, French drains, etc.). Most of these measures can deal with 75 to 85% of the rainfall events, but provisions still need to be made for extreme storms which will likely generate runoff that needs to be transported to neighbourhood- or watershed-scale stormwater systems.

Innovative Stormwater Management at the Neighbourhood Scale.

The innovative approaches to dealing with stormwater management within neighbourhoods are summarized in Table 2 and focus on street and parking lot runoff management.

Conventional streets in single family neighbourhoods are usually excessively wide and consist of curb and gutter systems that direct runoff into storm sewer pipes. In innovative approaches, curbs are removed and the street runoff is allowed to drain into swales incorporating sand filters. These swales are connected to detention ponds and wetlands in order to detain water from large storm events and to allow sediments and pollutants to settle in ponds; alternatively, runoff is allowed to infiltrate into the ground, where soil organisms and plants are capable of reducing the pollution effects. Wetlands are the most effective storage and filtration systems, but there are many safety and health concerns associated with developing constructed wetlands in the city.

The issues of stormwater quality enhancement at the neighbourhood level were addressed in four papers dealing with street sweeping, stormwater detention in stormwater management ponds, and stormwater biofiltration by compost filters.

The effectiveness of street cleaning as a source control measure reducing stormwater pollution is described in detail by Rochfort et al. (2009) in a paper entitled *Street Sweeping as a Method of Source Control for Urban Stormwater Pollution.*

The effectiveness of detention systems in reducing

TABLE 2. Innovative approaches to stormwater management at the neighbourhood level.

Traditional Approach	*Innovative Approach*
1. Pave all roads and sidewalks and direct runoff into storm sewers using a curb and gutter system	1. Minimize the width (area) of roads, remove all curbs and gutters and direct runoff into road-side infiltration swales, use pervious pavement (as much as possible)
2. Build a network of storm sewers and direct stormwater runoff into local streams	2. Build stormwater detention ponds and wetlands for large storms to detain runoff and reduce pollutant and sediment loads that enter streams
3. Build parking lots that are impervious and direct the runoff into storm sewers	3. Build parking lots with pervious pavement materials or direct runoff away from storm sewers into detention systems, swales and constructed wetlands
4. Allow contaminants to accumulate on street surface and be washed off by runoff into stormwater conveyance systems	4. Apply source controls by minimizing the use of polluting chemicals, and practicing street sweeping, contaminant retention, and rehabilitation of contaminated areas.

contaminants that can enter urban streams is described in a paper by Brydon et al. (2009) on *Evaluation of Mitigation Methods to Manage Contaminant Transfer in Urban Watersheds*. Detention systems promote stormwater settling, and the resulting sediment deposits need to be assessed, removed, and properly disposed of. The underlying environmental considerations for such maintenance operations are discussed by Westerbeek-Vopicka (2009) in her paper on *Sediment Assessment of Stormwater Retention Ponds within the Urban Environment of Calgary, Canada*.

Another method of enhancing runoff quality is by filtration through various media. The treatment of construction site runoff, with high concentrations of suspended solids, by compost filters was assessed by Taleban et al. (2009), as described in their paper on *Stormwater Runoff Treatment Using Compost Biofilters*.

One of the key public health concerns about stormwater ponds and constructed wetlands serving to detain and absorb contaminants is the problem of mosquitoes and the spread of West Nile virus. While the insect risk cannot be completely eliminated, the problem can be significantly reduced by a number of measures described in the paper by Jackson et al. (2009) entitled *Culex Mosquitoes, West Nile Virus, and the Application of Innovative Management in the Design and Management of Stormwater Retention Ponds in Canada*.

Another emerging stormwater management problem not discussed in this issue is the use of road salt in winter road maintenance and the resulting impacts on the environment. Road salts may interfere with operation of stormwater management facilities (e.g., by accumulation of chloride in stormwater ponds at toxic levels, release of chemicals from bottom sediments, impeded vertical mixing) and environmental impacts of road salts may be exacerbated by stormwater infiltration facilities diverting salt-laden stormwater into groundwater aquifers (Marsalek 2003). Warmer winters of recent years contributed to more fluctuation around the freezing point, and possibly a greater use of salt in road maintenance. Chloride saturated road runoff and chloride release from snow storage areas can have significant impacts on the aquatic ecosystem (Exall et al. 2008).

Innovation at the Watershed Scale

The individual stormwater management systems developed at the neighbourhood scale need to be integrated into a comprehensive drainage system within the watershed. A summary of the watershed-scale innovations that minimize drainage impacts is provided in Table 3.

Of foremost importance is the creation of a wide riparian buffer zone, also referred to as an ecological buffer (Day et al. 2008). In order to ensure effective functioning of buffer zones, it is required to design them sufficiently wide (a minimum width of 50 to 60 m) and with a good vegetative cover so that they uptake excess nutrients and provide sufficient space for absorbing and detaining sediments and pollutants, and for storing and detaining excess water during storm events. In many cities, the current buffer zone regulations require zone widths of 30 m, which is not sufficient to deal with all the impacts and the anticipated increases in storm event rainfalls and intensities. Ideally, a variable width buffer zone, depending on the local topography, surficial materials, and the hydrological regime, should be considered.

The second important aspect is to avoid channelization of urban stream and river beds. Hardened constructed channels may allow faster drainage of stormwater and prevent local bank erosion, but they often increase the risk of flooding in downstream areas because of restrictions imposed on lateral flows and storage outside of the stream channel. Maintaining natural channels should be given priority in all new urban developments, and conversion of constructed channels into more natural channels should be an important rehabilitation activity in old and established urban areas.

TABLE 3. Innovative stormwater management at the watershed scale

Traditional Approach	*Innovative Approach*
1. Stormwater is conveyed through pipes, passes through riparian buffer zones and is released into local streams	1. Create wide riparian buffer zones and create constructed wetlands within these zones to store excess stormwater, retain sediments and pollutants, and filter the water. Minimize or eliminate all stormwater outfalls discharging directly into streams
2. Channelizing urban streams and rivers to increase flow capacity, minimize bank erosion and speed up drainage	2. Maintaining natural river channels to allow lateral flow and storage of stormwater within the riparian zone
3. Floodplain is designated and flood management (protective) structures are built	3. Designate areas within the floodplain and the riparian buffer zone to serve for temporal storage of stormwater during flood events
4. All stormwater systems are connected and their outlets become point sources of pollution discharged into local streams	4. Avoid cumulative effects that increase flow and pollution loads by directing all stormwater drainage to pass through infiltration and detention systems

Floodplain management should also be changed because in the past most efforts have gone into building flood management (protective) structures (dikes, dams) and regulating land use activities. In view of increasing variability of climate, it is anticipated that flood events in cities will become more frequent and as a result, certain low-lying areas should be reserved for deliberate flooding. The land use should be such that activities can readily be restored after the floodwater has receded. Parks, wetlands and riparian buffer zones are the most appropriate sites for such considerations.

Finally, having many storm sewers conveying runoff directly into local urban streams results in concentration of flows as well as pollution. Retaining and infiltrating as much of the rainwater as possible, preventing generation of surface runoff, and detaining storm event runoff in ponds and wetlands within the watershed is the best strategy for adaptation to the increasing storm events and the increasing pollution from increased traffic and urban activities. No single measure will be sufficient to fully manage the aforementioned risks, but having a combination of these innovative measures in place should reduce the risk of flooding and the pollution associated with urban activities.

Integration and Effectiveness of Innovative Approaches

Using single innovation methods in isolation is only marginally effective, but to successfully reduce the anticipated increased risks of flooding and pollution requires a combination of innovative measures. Innovative initiatives also need to be linked within the watershed. The (Canadian) Water Balance Model (WBM) which was developed by an interprovincial body of experts (British Columbia Inter-Governmental Partnership 2003) is a first step towards facilitating integration of innovative stormwater management measures. It allows property owners to calculate runoff from impervious surfaces on their property and to determine the fraction of the rainfall depth that needs to be managed on site to maintain the onsite water balance. Once this is determined, a range of innovative options serving to detain and infiltrate the calculated rainfall/runoff can be identified.

Like any other hydrological model, WBM also has some limitations. The groundwater component and water table fluctuations are not sufficiently considered in the model. However, as a first approximation this WBM-based approach has a significant potential. Once the hydrological property assessment is completed, then there is a need to link the hydrology of the property to that of the neighbourhood and of the watershed. This can now be accomplished by using the QUALHYMO engine (Rowney and McRae 1991a, b) that was recently added to the WBM. As expected, the complexity of scaling the model results from the site to the watershed is challenging and examples of such attempts are not yet fully available. However, the basic idea of using spatially based models that can link the hydrological dynamics at the property scale with the cumulative effects over the neighbourhood and the watershed is a direction that has significant merit.

Some of the innovations discussed in this issue have been introduced in other countries, but have not been rigorously tested in different environmental regions of Canada. Many individual detention and infiltration systems have been constructed, but few are monitored to determine how effective they are in minimizing floods and reducing pollutants. A significant effort is needed to provide sound short- and long-term monitoring data so that one can calibrate the effectiveness of these measures. Only then we can recommend with confidence the use of the innovative methods elsewhere in new urban developments.

More efforts are also needed to determine the cost of constructing and maintaining innovative systems. Such a cost analysis should include comparison between conventional and innovative systems. This information is urgently needed in order to persuade developers to adapt these techniques. It is generally agreed that many of the innovative approaches are less expensive to build than conventional systems, but many of the long-term comparisons of maintenance costs of conventional and innovative systems are still lacking. In other words, comparisons of life-cycle costs between conventional and innovative stormwater management approaches are still missing. While it is possible to calculate flood damage costs, valorization of environmental benefits, such as improved aquatic ecosystem health, is most challenging.

Finally, given the widespread evidence that increased climatic variability and increased densification of urban areas are taking place simultaneously, it is evident that adoption of new approaches to stormwater management is urgently needed. Since we cannot anticipate what some of these cumulative effects will be and where the resilience and threshold levels are within the systems, it is prudent to use a multibarrier approach that includes a range of options that will likely differ from city to city depending on the topographic, geological, climatic, land use, and environmental conditions. At the same time, there are many obstacles to innovation in stormwater management, including the prevailing legal environment that focuses on litigation, and the many overlapping fragmented jurisdictions that make approvals of development permits complex and time consuming. Often, the additional requirements associated with moving from the conventional to the innovative approach are excessive and discourage the innovation. What is needed are incentives to facilitate and speed up the approval process and to provide experimental permits that allow developers to experiment. The key is to promote well documented case studies demonstrating innovative stormwater management that is effective in reducing the risk of flooding and pollution, and at the same time is more cost effective, than the conventional approaches, over the long run.

Conclusions

This special journal issue on innovative stormwater management brought together a wide range of Canadian experts who specialize in all aspects of urban stormwater management in various parts of Canada. The focus of the featured papers is on introducing innovative approaches that range from rainwater retention, reuse, and infiltration at the property level, to detention and infiltration of runoff at the neighbourhood scale. All such efforts should take into account the linkages to the urban watershed by managing large riparian buffer zones, refraining from channelization of streams and rivers, and designating areas in floodplains for temporary storage of stormwater during storm events. Roof-water harvesting, minimization of impervious surface areas, diversion of runoff into detention and wetland systems, use of pervious pavements, and directing road and parking lot runoff into filter systems are all approaches that are well documented in different parts of Canada. Increasing health concerns about insects and the spread of disease in areas adjacent to urban ponds and wetlands, and impacts of road salts on streams, as well as economic considerations, are all addressed in this introduction. The recommended approach emphasizes the use of a wide range of combinations of innovative measures, rather than focusing on single innovations, in order to hedge against uncertainties and to integrate individual efforts between the site, neighbourhood, and watershed levels.

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Evaluation of Mitigation Methods to Manage Contaminant Transfer in Urban Watersheds

Julia Brydon, Iwata Oh, Julie Wilson, Ken Hall, and Hans Schreier*

Institute for Resources and Environment, The University of British Columbia, 4th floor, 2202 Main Mall, Vancouver, B.C. V6T 1Z4

Three case studies on trace metal contamination in urban stormwater are presented from the Greater Vancouver area of British Columbia. In the first case study, the spatial and temporal variability in trace metals in sediments were determined in the completely urbanized Brunette watershed. A natural lake in the middle of the watershed acts as a sediment detention system, and an analysis of the sediment core showed the historic accumulation of metal and selective organic contaminants in sediments since the early 1800s. Suspended sediments transported during storm events showed significantly higher concentrations of trace metals than bedload sediments, and the largest proportion of the geochemically active metals was found to be associated with the organic-sulphur-based fraction. Benthic organism survival tests showed mixed results with lower survival and growth in urban sediments than in control sediments from a forested watershed. In the second case study, significant correlations were obtained between percent impervious cover and trace metal concentrations in 28 subwatersheds with various degrees of urbanization. It is shown that imperviousness combined with traffic density can significantly improve the prediction of metal contamination in highly urbanized watersheds. In the third case study five urban stormwater detention systems were examined over one year to determine how effective these systems were in removing metal contamination. The results were highly variable depending on a wide range of physical conditions, land use activities, traffic volume, and detention system designs. The range of total metal detention was between −15 to +72% for copper (Cu), lead (Pb), and zinc (Zn), while iron (Fe) and manganese (Mn) retention was generally poor. Labile Zn was more effectively retained in four of the five ponds, and a significant relationship was found between percent imperviousness, traffic volume, and Zn concentrations in water, sediment, and labile form.

Key words: **imperviousness, trace metals, sediment quality, cumulative effect, diffusive gradients in thin films (DGT), detention ponds**

Introduction

The extent of impervious surface coverage in urban watersheds is commonly used as a cumulative effects indicator (Schueler 1994; Booth and Jackson 1997). Imperviousness is the indirect cause of urban water pollution and its impact on aquatic organisms. Imperviousness increases surface runoff, causes flashy streamflow responses, increases erosion, and transfers contaminants that accumulate on road surfaces and driveways into receiving waters. Depending on the antecedent rainfall/runoff relationship, higher contaminant loadings and rapidly changing concentrations of pollutants during storm events are associated with increasing impervious cover (Larkin and Hall 1998; Ichiki et al. 2001). Relationships between imperviousness and aquatic health have, in most cases, shown negative curvilinear relationships between the percent of imperviousness and benthic macroinvertebrate diversity (May et al. 1997; Stepenuck et al. 2002; Wang and Kanehl 2003; Novotny et al. 2005). The literature suggests that if the effective impervious surface areas (impervious surfaces directly connected via a drainage system to urban streams) are less than 10 to 12%, it is possible to maintain a diverse and healthy aquatic community and minimize habitat degradation (Cianfrani et al. 2006). Catchments containing 12 to 25% impervious surfaces are considered impacted, and those above 25% are considered unhealthy aquatic environments (Cianfrani et al. 2006). Although total impervious area can easily be determined from aerial imagery, it is not a direct measure of the effective impervious area, which can only be determined by costly and time-consuming field investigations (Cianfrani et al. 2006). The underlying factors that determine the impact of imperviousness on hydrologic regime, water quality, and aquatic health of a stream are complex. The wet weather performance of different best management practices is highly variable in different geographic areas (Booth et al. 2002; Pennington et al. 2003; Pontier et al. 2004; Novotny et al. 2005). Accordingly, two research questions are emerging from the literature: 1) Can we develop new urban watersheds with less than 8 to 12% impervious surface cover? 2) What are the best options to mitigate impacts on streams in urban watersheds, particularly those with >12% impervious surface cover?

The main sources of contamination in urban watersheds are trace metals, organic contaminants, nutrients, pathogens, and dissolved solids (Sartor and Boyd 1972; Stone and Marsalek 1996; Deletic and Orr 2005) that accumulate on road surfaces from car traffic and from urban land use activities. These contaminants enter receiving waters in both dissolved and particulate forms and are transported primarily during storm events. Contaminant transfer is highly variable and dependent on land use activities, traffic density, storm intensity, storm duration, and the conveyance system that links impervious surfaces to the stream channel. The objectives of this paper are to 1) quantify sediment-associated pollutant levels in the highly urbanized Brunette watershed in the Vancouver area, 2) examine the effect of imperviousness and traffic density on contaminant loading of metals, on sediments, and on aquatic health, and 3) determine the efficiency of retaining metals in detention systems and wetlands. Three case studies are presented to meet the proposed objectives.

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* Corresponding author: star@interchange.ubc.ca

Case Study Materials and Methods

Case Study 1: Examining Sediments in the Brunette Watershed

The Brunette watershed is a highly urbanized watershed in the Burnaby-Vancouver area, which has approximately 45% impervious cover and has two of the busiest transportation corridors in the Lower Fraser Valley. The peak a.m. traffic volumes are 5,000 to 7,000 vehicles per hour on the Trans-Canada highway and 3,000 to 5,000 vehicles per hour on the Lougheed Highway (McCallum 1995) (Fig. 1). Burnaby Lake is located in the middle of the Brunette watershed and is a natural sediment trap. Concerns have emerged about sedimentation in the lake, the loss of recreational opportunities, and related impacts on aquatic and human health (GVRD 2001). Over the past 35 years, several studies have been conducted in the Brunette watershed to determine levels of trace metal contaminants in bed-sediments, in lake core samples and suspended sediments in tributary inflows, in the lake outflow, and at locations downstream (Bindra and Hall 1977; McCallum 1995; Muraro 2005; Li 2007).

Five sample sites were selected in the Brunette Watershed (Fig.1). Still Creek and Eagle Creek are two tributaries draining into Burnaby Lake. Stoney Creek is a first-order tributary entering the river below the lake, and the Lower Brunette River station is below the lake which drains into the Fraser River (Fig. 1). The land use in each subwatershed is described in Table 1.

Bed sediments were collected in late summer of 1973, 1993, and 2003. The samples were wet sieved and the <63-μm sediment fraction (Fergusson 1990) was analyzed using the hydrofluoric acid digestion (HF) method (Bindra and Hall 1977; McCallum 1995; Muraro 2005).

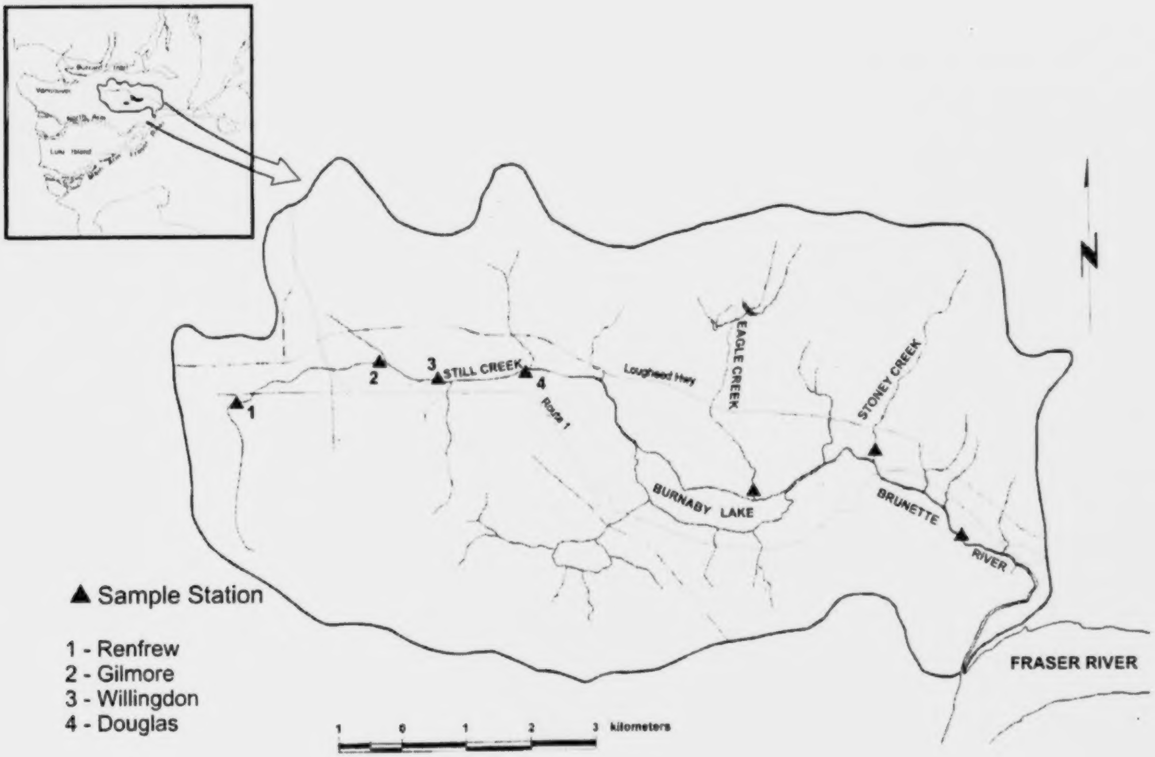

Fig. 1. Location of sampling stations in the Brunette watershed.

TABLE 1. Population, traffic density, and imperviousness in main subwatersheds of the Brunette River

Subwatershed	*Population (total #)*	*Population Density (#/km²)*	*Peak hour vehicle traffic (vehicle/km²/hr)*	*TIA[a] in subwatershed (%)*		
	2001	*2001*	*2001*	*1973*	*1993*	*2000*
Still Creek	135,154	6,115	165,000	47	54	68
Eagle Creek	19,232	3,260	20,000	16	27	30
Stoney Creek	11,530	1,624	25,000	19	27	32
Lower Brunette	32,956	3,682	37,000	34[b]	41[b]	45[b]

[a] TIA = total impervious area.
[b] Includes entire watershed

The Pb, Zn, Cu, Mn, Ni, Cr, Fe, and Co concentrations were determined using the atomic absorption and inductively coupled plasma atomic emission spectroscopy (ICP-AES) instruments. The bioavailable metal fractions were determined from sediment samples from three stations in Still Creek and one site in Eagle Creek. Various geochemical phases were determined using the sequential extraction procedure described by Engler et al. (1974) and Rule and Alden (1992).

The growth and survival of *Chironomus riparius* and survival of *Hyalella azteca* were assessed to estimate sediment toxicity using standard bioassays (Ingersoll and Nelson 1990). The solid phase 5-minute Microtox EC50 test which measures decreases in light emission of photoluminescent bacteria was also used to measure sediment toxicity (Microbics Corp. 1991).

In 2004, three 1.5-m long sediment cores were obtained near the inlet of the lake, and the cores were examined to document historic changes in sedimentation rates and contaminant accumulation over time. The Pb^{210} dating technique (Graney and Eriksen 2004) was used on the best core, and selective sediment layers were dated and analyzed for trace metals and chlorinated organics. The HF technique described by McCallum (1995) was used to determine the metal concentrations, and the chlorinated hydrocarbons (dichloro-diphenyl-trichloroethanes [DDTs] and polychlorinated biphenyls [PCBs]) were extracted from the sediments with acetone and then partitioned into hexane. Individual compounds were detected using electron capture gas chromatography (McCallum and Hall 1998).

A third study examined the metal loadings to the lake during several storm events using a field-based centrifuge. The suspended sediment was analyzed using techniques described by Sekela et al. (1998).

Collectively, these data provide important information about the effectiveness of the lake as a natural detention system, and illustrate the spatial and temporal variability in metal contamination present.

Case Study 2: Relationships Between Imperviousness, Traffic, and Metal Pollution in Sediments

Twenty-eight subwatersheds in the Greater Vancouver area were examined to determine the effects of imperviousness and traffic density on sediment contamination. Bed sediments were sampled in 1973 (Westwater Research Centre 1973) and again in 2006, and the changes in imperviousness were determined from historic aerial photos and recent orthophotos to measure changes in metal concentrations as a function of the increasing imperviousness at the regional scale over time (Iwata 2007). For the 2006 samples, the <63-µm fraction and the U.S. EPA (1994) extraction method combined with the ICP-AES instrument were used for the analysis. The 1973 sediment samples were analyzed with the aqua regia-HF digestion method (Smoley 1992) followed by absorption spectrophotometric analysis instead of ICP-AES (Iwata 2007). Since the analytical methods have improved dramatically over this 33-year time period, it is difficult to compare the changes in metal concentrations between the two data sets. Only the very large changes in concentrations are likely relevant. The change in imperviousness from 1973 to 2006 was determined using digital orthophotos of the area and geocorrected digital aerial photos. The changes in imperviousness from 1973 to 2006 were determined using geographic information system (GIS) overlay techniques.

Case Study 3: Effectiveness of Metal Retention in Stormwater Ponds and Wetlands

The third case study was designed to demonstrate the effectiveness of five constructed wet and dry detention ponds and wetlands in detaining metals in sediments from urban runoff. The detention systems were selected in the Greater Vancouver and Lower Fraser Valley areas, and the input and outflow of sediments and associated trace metals were determined over a one-year period (Brydon 2004).

Two of the five study ponds were built primarily for the purpose of peak runoff control (Westview and 52 Ave/221A St.), while three also incorporated additional measures for water quality improvement (Tempe Heights, Griffin Park, and Oakalla). The ponds represented a cross section of the attempts to implement stormwater ponds in the area since the early 1990s. They are all located in suburban areas, four of which collect runoff from housing developments and one which collects highway runoff (Westview). The catchments vary in size, traffic intensity, and percent impervious cover. The pond designs range from a strict detention pond to a constructed wetland, with varying surface areas, volumes, and extents of vegetation. Although these factors have been shown to affect metal removal (Mallin et al. 2002; Walker and Hurl 2002; Minton 2005), they provide an opportunity to observe the level of contamination in the sediments and water in different systems. A summary of the characteristics of the individual ponds is provided in Table 2.

The monitoring program consisted of analyzing accumulated sediments in the detention systems over a one year period (2003 to 2004) and exploring the use of diffusive gradients in thin films technique (DGT) to measure labile metal concentrations in the water. The DGT units were deployed for periods of three to four weeks at the inlet and outlet of each pond and wetland. The DGT units consisted of a binding agent that accumulated solutes quantitatively after their passage through a well-defined diffusion layer. A polyacrylamide hydrogel was used as the diffusive layer, while Chelex 100 resin, incorporated into a second gel layer, served as the binding agent. The two gels were enclosed in a small plastic device that could be deployed in situ (Zhang et al. 1998; Gimpel et al. 2001; Peters et al. 2003; Zhang 2003). Since ion exchange resins accumulate ions through exchange mechanisms, they model the action of cell walls on plant roots, fish gills, and aquatic invertebrates (Reichman 2002).

DGT preconcentrates metal solutes, which alleviates the problems of contamination and poor detection at trace levels. DGT units have proven to be useful due to their ability to operate in situ and eliminate the need for sample collection and storage. It has been found that the concentrations of metals measured by DGT in solutions with pH values between 5 and 8.3 agree well with those

TABLE 2 . Characteristics of selected detention ponds

	Westview interchange detention pond	*52Ave/221A Street detention facility*	*Tempe Heights pond*	*Griffin Park biofiltration pond*	*Oakalla biofiltration system*
Location	City of North Vancouver	Langley	City of North Vancouver	District of North Vancouver	Burnaby
Date of completion	1996	1999	1993	1998	1990
Reason for construction	Detention of stormwater runoff to reduce volume entering Mosquito Creek	Detention of stormwater runoff to reduce volume entering the Nicomekl River	Detention of stormwater runoff to reduce volume; improve water quality entering Keith Creek	Detention of stormwater runoff to reduce volume; improve water quality entering Mosquito Creek Salmon Habitat Area	Detention of stormwater runoff to reduce volume; improve water quality entering Deer Lake
Catchment area (CA) (km ^{2})	2	0.28	0.0014	0.0036	0.3
Catchment land use	Major highway	Residential subdivisions, commercial	Residential subdivisions and park	Residential subdivisions and park	Residential subdivisions
Percent impervious cover (%)	100	50	65	60	55%
Surface area (SA) (m ^{2})	300	4000	608	90.3	966
CA/SA	6666.7	70	2.3	39.9	310.6
Shape	Oval with curve after inlet	2 square cells divided by bridge with culvert	Oval with curve near middle of pond	Meandering channel	Two ponds and one large marsh
Volume (m ^{3})	300	4,900	668.8	72.24	1,615
Percent vegetation (%)	0	50	20	60	70
Types of vegetation	None	Primarily cattails (*Typha*)	Cattails (*Typha*)	Cattails (*Typha*); native plants, grasses	Cattails (*Typha*); grasses
Average annual daily traffic (year collected)	72,000 (2000)	900 (2000)	960 (2004)	7,108 (2000–2002)	640 (2001)

obtained by direct measurement. When DGT is deployed in rivers or streams with reasonable flow rates, the measurement should be independent of flow to within 5% (Zhang et al. 1998; Gimpel et al. 2001).

Bed sediment sampling took place in July 2003 and again in September 2003 followed by systematic sampling every three to four weeks from November 2003 to January 2004 in response to frequent rainfall. As precipitation diminished, sediment sampling was reduced to six to seven week intervals from January to June 2004. Surficial sediments were collected at the inlet and outlet of each pond and wetland.

The DGT units were transported to the laboratory in small plastic bags and analyzed according to directions provided by Zhang (2003). The resin gel layer was shaken in a 30-mL plastic Nalgene bottle containing 20 mL of 1 M nitric acid. The resin gel was left in the acid for 72 to 96 hours to ensure that sufficient elution time was provided. The nitric acid extracts were analyzed using ICP-AES. The concentrations reported for DGT results represent the concentration of metals measured in the eluent. Sediment samples were wet sieved to the <63-μm fraction, were digested using the aqua regia technique, and were analyzed for metals using ICP-AES (Smoley 1992).

Quality assurance/quality control procedures for all analyses included field and laboratory duplicates and blanks. Sediment analysis also included the use of standard reference sediment, and the DGT units were calibrated in the laboratory prior to deployment according to guidelines provided by the manufacturer (Zhang 2003).

Results

Case Study 1

Results of metal concentrations in sediments in the Brunette watershed. The results from the bed sediment analysis showed that the dry weight concentrations of Cu, Pb, and Zn were significantly higher in Still Creek than in the lower Brunette River (Fig. 2). This trend appears to be the result of Burnaby Lake acting as a sink for sediment and associated metals. There has been a general decline in some metal levels over time (particularly Pb concentrations) since source control measures, such as removal of Pb from gasoline, were introduced in the 1980s. The extent of impervious surface area and related traffic volume is likely the main cause of the fairly constant concentrations of Zn in all stations and of Cu in Eagle Creek and the Brunette River. The high concentration of Mn in the Eagle Creek subwatershed is likely due to direct anthropogenic inputs from eroded clay-rich deposits of marine origin that were exposed by construction in the headwaters of the creek.

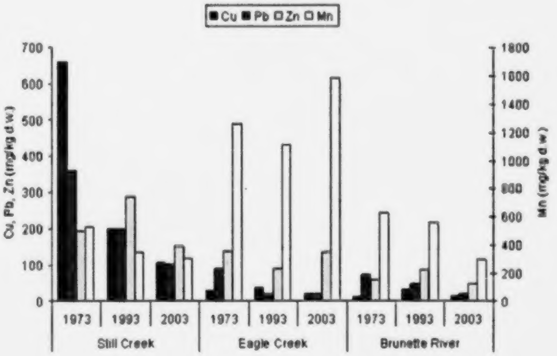

Fig. 2. Trace metals in bed sediments in the Brunette watershed, 1973 to 2003.

Results of a historic reconstruction of accumulated contaminants in sediment core from Burnaby Lake. A historic profile of metal contaminant accumulation in the sediment core is shown in Fig. 3. There was a steady increase in Pb concentration from the 1920s until the early 1980s when source controls, such as Pb removal in gasoline, were implemented. The observed decrease in Pb during the 1970s resulted from the accumulation of a large sand layer likely deposited during the largest rainfall event recorded over the past 100 years. In contrast, Zn and Cu concentrations continued to accumulate and this is attributed to the increase in traffic over time.

Accumulation of organic pollutants such as PCBs and DDT in the Burnaby Lake sediment core occurred from the 1950s to the 1980s (Fig. 4). In shallower sediments, these concentrations declined rapidly after DDT was banned in Canada and uses of PCBs were prohibited. Presently, 360,000 m^3 of sediment are scheduled to be dredged from Burnaby Lake by 2012 to improve water quality and recreational opportunities (BCEAO 2008). There is a serious concern of resuspending contaminants during the reclamation process and dredging, and onsite treatment must be carried out before the sediments can safely be disposed of in landfills.

Trace metals in suspended sediments during storm events. Using suspended sediment data allowed the determination of annual metal loading to Burnaby Lake. The results shown in Table 3 indicate the contribution to the annual loading of every hectare of urban area in the subwatersheds. As was determined by Sekela et al. (1998) and Li (2007), between 50 to 750 g of Cu, Pb, and Zn is estimated to originate each year from every hectare of urban land in the watershed.

Similarly, high concentrations of organic pollutants, such as polycyclic aromatic hydrocarbons (PAHs), PCBs, and derivative byproducts of DDT, were transported during this storm event and the loading rates were generally twice as high at the Burnaby Lake intake (Still Creek) than in the Lower Brunette River below the lake (Table 4). This again suggests that sediments are trapped in the lake, which results in a significant reduction in downstream concentrations of these contaminants.

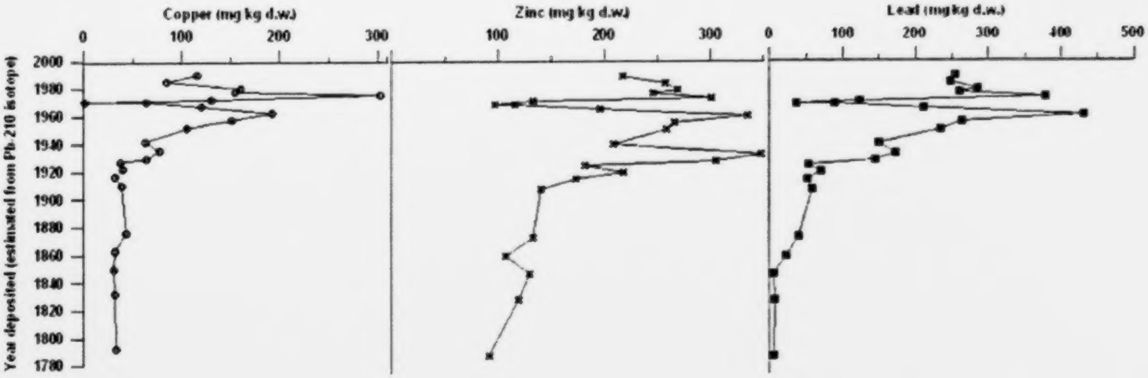

Fig. 3. Historic records of Cu, Zn, and Pb concentrations in sediment cores from Burnaby Lake (d.w. = dry weight).

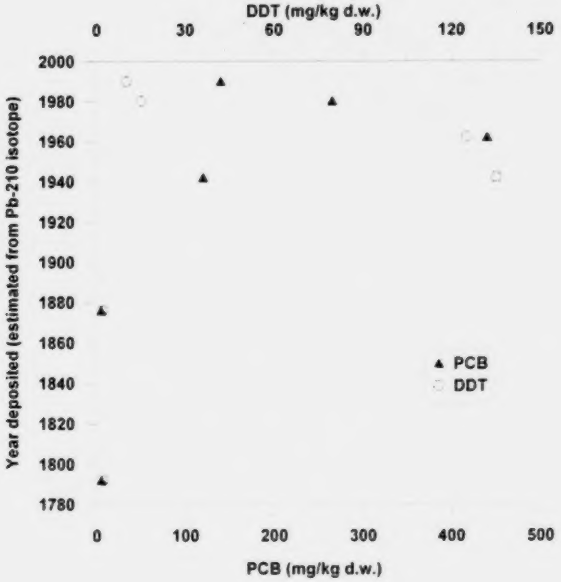

Fig. 4. Historic records of DDT and PCB concentrations in sediment cores from Burnaby Lake (d.w. = dry weight).

TABLE 3. Trace metals in suspended sediments in the Brunette Watershed

Subwatershed	*Metals in suspended sediments 8-h event (1997) (mg/kg [dry weight])*			*Total metals transported during event (1997)* [a] *(kg/event)*			*Loadings of metals in suspended sediment s (2003)* [b] *(g/ha/year)*		
	Cu	*Pb*	*Zn*	*Cu*	*Pb*	*Zn*	*Cu*	*Mn*	*Zn*
Still Creek	320	254	772	1.320	0.712	3.896	105	600	180
Eagle Creek							50	500	160
Stoney Creek [c]							135	750	255
Brunette R.	164	175	557	0.656	0.424	2.168	125	450	85

[a] Sekela et al. 1998 .

[b] Li 2007.

[c] Higher concentrations in Stoney Creek could be the result of major construction activities at Simon Fraser University in 2003 exposing clay rich marine sediments.

TABLE 4. Organic contaminants in suspended sediments in the Brunette watershed[a]

Subwatershed	*Selected organic contaminant concentrations*[b] *(8-h event 1997)*				*Total organics (over 8-h event 1997)*	
	LPAH[c] *(μg/kg)*	*HPAH*[d] *(μg/kg)*	*PCB (μg/kg)*	*DDT (μg/kg)*	*DDT-t*[e] *(μg/event)*	*PCB (mg/event)*
Still Creek	3,210	25,370	272	53.7	209	1,010
Brunette R.	1,744	13,980	106	37.2	46	550

[a] Sekela et al. 1998.

[b] Units for all four parameters in full are μg/kg dry weight in suspended sediments.

[c] LPAH = Low molecular weight PAHs.

[d] HPAH = High molecular weight PAHs.

[e] DDT-t = total "o" and "p" isomers of DDT and degradation products dichloro-diphenyl-dichloroethylene (DDE) and dichloro-diphenyl-dichloroethane (DDD).

Bioavailability and toxicity of metals. Maintaining a healthy aquatic ecosystem in urban watersheds is a management priority, and measures are needed to reduce the fraction of bioavailable metals entering receiving waters. The data in Fig. 5 show that Cu was primarily associated in the organic-sulphur phase (OSP) which was proportionally higher (up to 50%) than all other fractions. Thirty to 50% of the Zn was associated with the easily reducible sediment phase in 1973, which decreased to 10 to 20% in 1997, with a higher proportion in the OSP in 1997. The highest contaminated station for both Cu and Zn in 1973 was in the lower reaches of Still Creek (Douglas Ave.), but this decreased by 1997, likely as a result of the phasing out of heavy industry in the area above the station.

With the exception of Cu in the Brunette River (Table 3), the levels of Cu, Pb, and Zn in the suspended sediments exceeded the probable effects levels (PEL) that have been considered by CCME (2003) to be detrimental to aquatic biota. Research on the speciation of trace metals in sediments substantiates the distribution of relatively high levels of Cu associated with the organic-sulfur phase, and of Zn with the easily reducible sediment phase (Fytiano and Louranton 2004; Wang et al. 2004; Van der Veen et al. 2006; Morillo et al. 2007).

Table 5 shows *Chironomid* and *Hyalella* survival and Microtox test results for 1996 and 1997 (Microbics Corp. 1991). No clear trends were observed during these tests. However, when the survival and growth test for *Chironomids* and *Daphnia* were compared with those from Musqueam Creek (an undisturbed creek in a forested environment) the impact of the urban pollution was clearly evident (Table 5) (Smith 1994).

The relationship between sediment contaminant levels and negative impacts on aquatic biota starts to become more significant when one considers both trace metals and organic contaminants relative to their cumulative PEL (Fig. 6). The x-axis of Fig. 6 represents the fractional sum of the trace metals (Cu, Pb, and Zn), each divided by their PEL in addition to the similar fractional sums of three PAHs (phenanthrene, fluorine, and pyrene) (equation 1) (CCME 2003).

$$CumulativePEL = \frac{Cu}{PEL_{Cu}} + \frac{Pb}{PEL_{Pb}} + \frac{Zn}{PEL_{Zn}} + \frac{PAH_1}{PEL_1} + \frac{PAH_2}{PEL_2} + \frac{PAH_3}{PEL_3} \quad (1)$$

Where: PAH_1 = phenanthrene, PAH_2 = fluorene, PAH_3 = pyrene.

Both the solid-phase Microtox (EC_{50}) and the percent survival of *Chironomids* in surface sediments from four stations in Still Creek (Fig. 1) demonstrated a toxic response in relation to the cumulative weighted levels of these contaminants. The results show that metal concentrations in urban sediments are highly variable

TABLE 5. Effect of sediment toxicity on aquatic biota at five sites in the Brunette watershed

		Chironomid survival (%)			*Chironomid growth (mg)*	*Hyalella survival (%)*		*Daphnia survival (%)*	*Microtox (5-min EC50)*	
		1996	*1997*	*2002*	*2002*	*1996*	*1997*	*2002*	*1996*	*1997*
Still Creek	Renfrew St.	90	82.5			20	100		NOE[a]	NOE
	Gilmore St.	90	95			97.5	95		14.2	NOE
	Douglas St.	100	85	20	1.4	90	100	70	8.7	NOE
Eagle Creek	Piper Ave.	97.5	75			100	100		NOE	16.4
Musqueam Creek	Control Site			77	2.05			90		

[a] NOE = no observable effect.

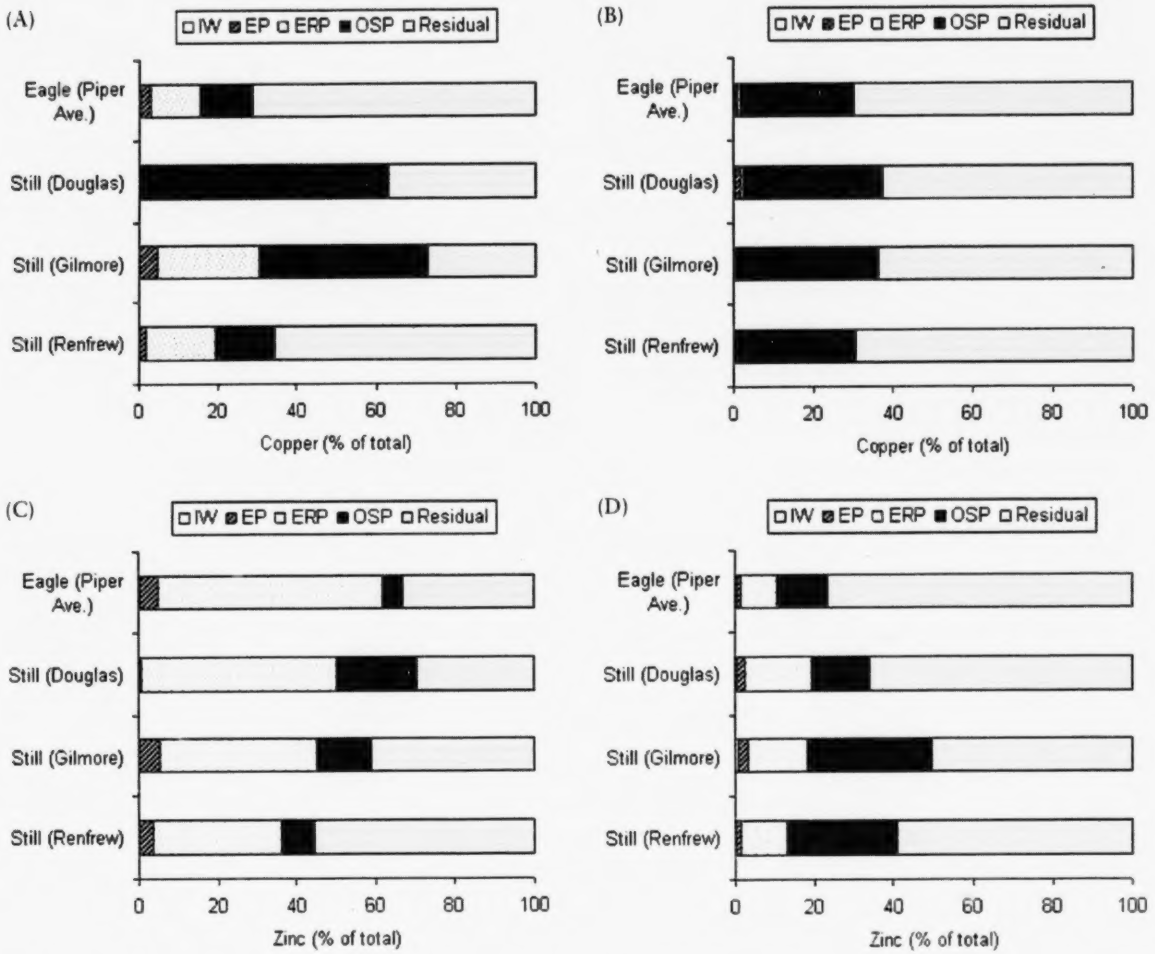

Fig. 5. Percent of total a) Cu (1973), b) Cu (1997), c) Zn (1973), and d) Zn (1997) concentrations found in different phases in Brunette River watershed sediments (<63 μm). IW = interstitial water; EP = exchangeable phase; ERP = easily reducible phase; OSP = organic sulphur phase.

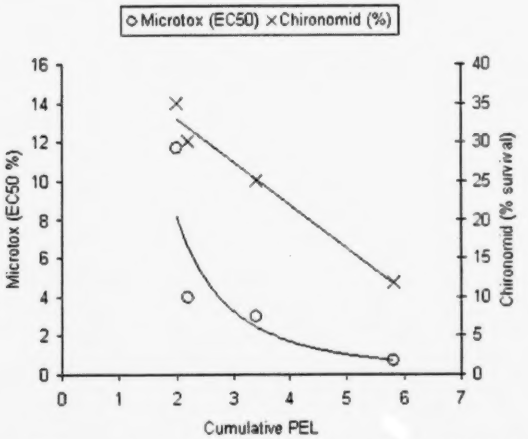

Fig. 6. Comparison of sediment quality guidelines (probable effects levels [PEL]) and bioassays.

and the concentrations in the suspended fractions, particularly during storm events, are significantly higher than in the bedload. The bioavailable fraction of trace metals will tend to be associated with the chemical fraction that is mobilized by weaker chemical reagents and can potentially move across cellular membranes and then become toxic to organisms. Other bioassays show sediments from a highly impacted urban site to be far more detrimental to the survival of test organisms than sediment from a control site unaffected by land use changes.

Case Study 2

Regional relationships between imperviousness, traffic, and metal pollution in sediments. The imperviousness increased between 1973 and 2006 in 23 of 24 subwatersheds with comparable data, but the changes

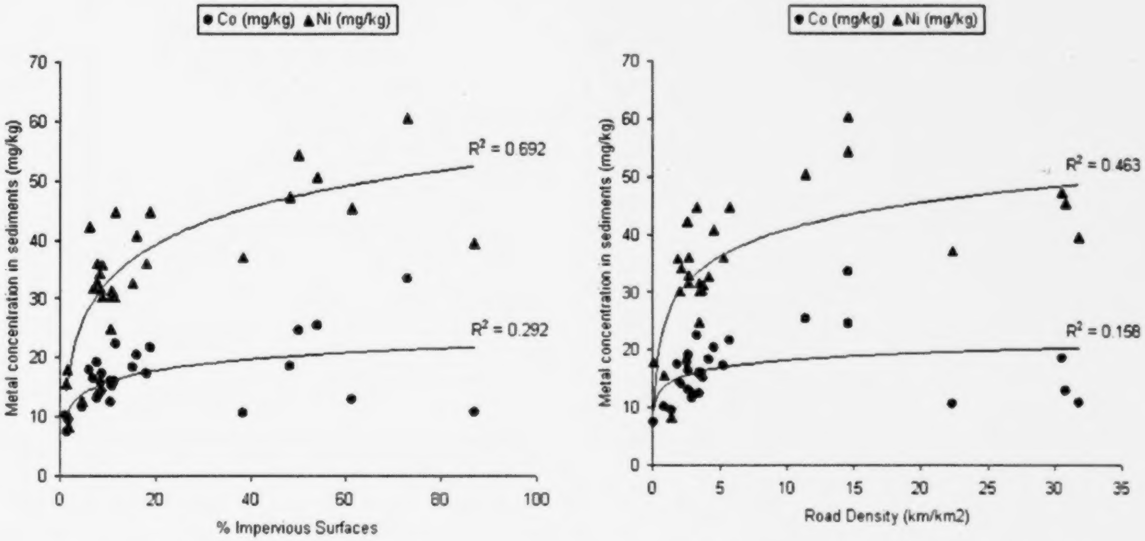

Fig. 7. Relationship between cobalt and nickel concentrations in sediments and a) percent imperviousness and b) road density in 28 subwatersheds. Significant logarithmic correlations ($p < 0.05$).

were relatively small (<9%) in 20 of these subwatersheds. In those subwatersheds where the changes in urbanization were large (>30%) it was evident that trace metals in sediment have also increased in a significant manner. The trace metal contamination in bed sediments increased between 30 and 50% in those watersheds that had more than a 30% increase in impervious surfaces between 1973 and 2006.

The 2006 results showed that there were significant logarithmic relationships between the percent imperviousness and the Ni and Co concentrations in sediments (Fig. 7). In contrast, Zn and Cu showed significant linear relationships (Fig. 8). Since the major sources of Zn and Cu originate from transportation activities, it was anticipated that road density rather than total imperviousness would provide a better relationship with these metals, but the relationships between these factors were insignificant (Table 6). It is likely the traffic volume and not the road density will have the greatest impact on metal accumulation in sediments.

Another way to refine the imperviousness index is to determine the effect of urbanization within a 100-m riparian buffer zone because land use activities within this zone are likely to have a more pronounced impact on the urban stream (Wang et al. 2001; Wang and Kanehl 2003). However, when imperviousness and road density were examined within the 100-m riparian buffer zone, these relationships did not improve significantly (Table 6).

These overall relationships between imperviousness and Cu, Zn, Co, and Ni are adequate as a first approximation, but it should be remembered that imperviousness as determined from aerial imagery is a crude indicator. To complement it, analysis should include a field assessment of the effective impervious area, the amount of traffic, other urban land use activities, and the prevailing hydrological regime. The four most urbanized subwatersheds were selected from West Vancouver and from the Alouette watershed, and were compared. They all have similar degrees of percent impervious areas; however, it is evident from Fig. 9 that this can generate very different Cu and Zn concentrations in bed sediments. Cu and Zn concentrations were significantly higher in the West Vancouver subwatersheds than in the Alouette. A possible explanation for this is that the West Vancouver subwatersheds are all transected by a major highway (estimated at 20,000 vehicles/day), while those in the Alouette system contain extensive road networks in suburban neighbourhoods with significantly lower traffic densities (estimated at 500 to 1,000 vehicles/day). Unfortunately, most low road density areas do not have traffic information in order to quantify such impacts.

TABLE 6. Significant correlations (R^2, $\alpha = 0.05$) between imperviousness, road density, and metals in sediments in 2006 ($n = 28$)

	Cu[a]	*Zn*[a]	*Ni*[b]	*Co*[b]
% Impervious surfaces	0.306	0.769	0.692	0.292
Road density	0.401	0.719	0.463	0.158
% Impervious surfaces in 100 m buffer zone	0.570	0.717	0.489	0.158
Road density in 100 m buffer zone	0.498	0.508	0.223	N/S[c]

[a] Cu and Zn follow linear nonparametric Spearman's rank correlations.

[b] Ni and Co follow logarithmic curve.

[c] N/S = not significant.

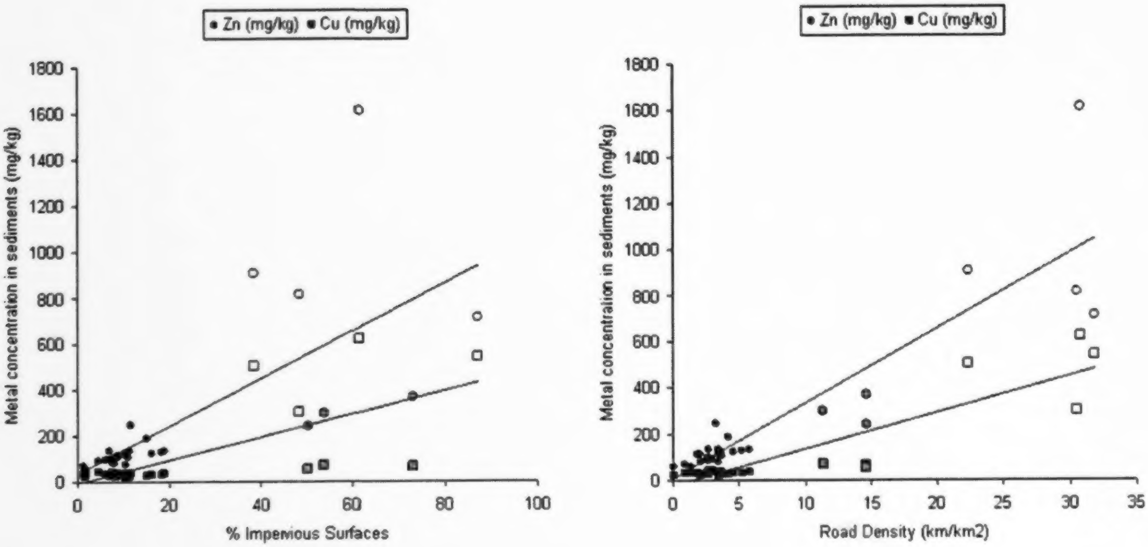

Fig. 8. Relation between zinc and copper in river sediments and a) percent imperviousness and b) road density in 28 subwatersheds. Grey symbols correspond to areas with very little traffic flow. Open symbols are points sampled near high traffic causeways. Black symbols represent low traffic in residential areas. Significant Spearman's correlations ($p < 0.01$).

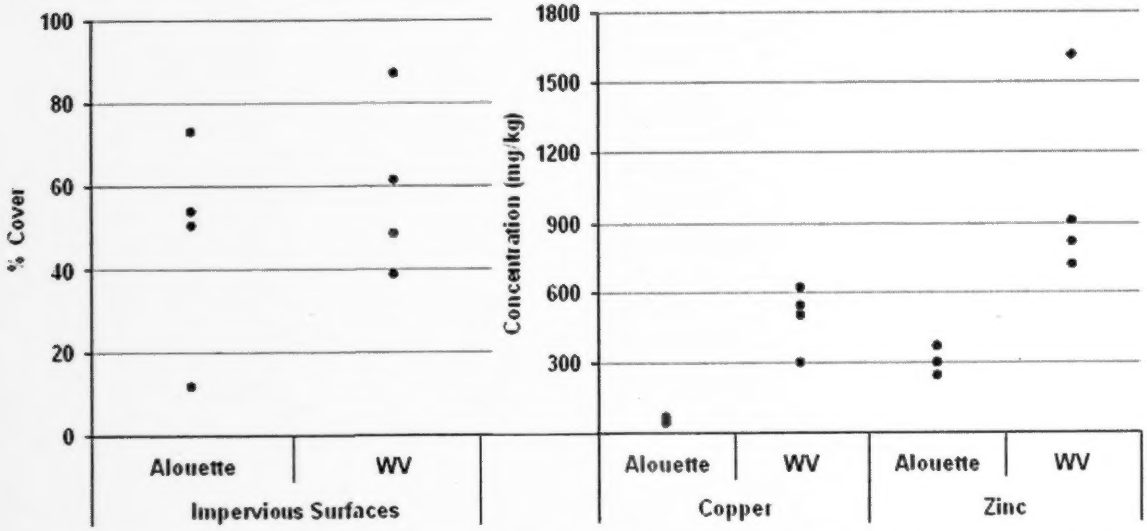

Fig. 9. Differences in imperviousness and Cu and Zn between four urbanized subwatersheds in West Vancouver (WV) and in the Alouette watershed (2006).

Further research is needed to investigate the cumulative effects of traffic density, road density, and impervious surface cover on urban water quality, especially during storm events.

Case Study 3

Effectiveness of metal retention in stormwater ponds and wetlands. Comparing metal levels in different ponds showed that the highest sediment contamination originated in the Westview catchment area, particularly with respect to chromium and zinc concentrations. Griffin Park and Tempe Heights exceeded the PEL in 100% of samples for inlet copper and zinc in the sediments (CCME 2003). The sediments at the Oakalla biofiltration system had significantly higher manganese concentrations than the other ponds.

Table 7 shows the highly variable differences in total trace metal concentrations in sediments between the inlet and outlet of each pond. The retention was generally more

TABLE 7. Total metals in sediments at five stormwater ponds in Greater Vancouver (median values) in wet and dry seasons [a]

	Cr		*Cu*		*Fe*		*Mn*		*Pb*		*Zn*	
	Wet	*Dry*	*Wet*	*Dry*	*Wet*	*Dry*	*Wet*	*Dry*	*Wet*	*Dry*	*Wet*	*Dry*
52Ave/221A Street Detention Facility (n = 4 [wet], 3 [dry])												
Inlet (mg/kg)	40.4	33.8	80.9	83.9	20127.8	23607.5	1333.7	1191.6	38.1	45.2	342.3	457.3
Outlet (mg/kg)	30.2	30.0	43.9	37.1	18385.5	19779.7	476.7	425.0	13.7	12.0	188.1	166.9
Percent difference (%)	29.40	21.00	52.34	62.26	12.99	16.21	60.02	72.86	66.63	72.34	50.66	63.51
Griffin Park Biofiltration Pond (n = 4 [wet], 4 [dry])												
Inlet (mg/kg)	30.4	38.2	258.1	278.6	14133.3	25482.9	194.3	578.3	157.8	237.4	669.0	666.8
Outlet (mg/kg)	16.1[b]	19.5[b]	84.5[b]	108.1[b]	22308.18	24941.6	299.5[b]	566.4	85.1[b]	154.1	230.1[b]	297.4[b]
Percent difference (%)	44.58	53.99	67.19	64.78	-34.30	-2.93	-38.04	5.06	49.11	26.79	65.45	51.81
Westview Interchange Detention Pond (n = 4 [wet], 4 [dry])												
Inlet (mg/kg)	57.5	81.3	392.7	642.1	12305.2	20402.2	164.4	274.6	199.2	218.3	1030.2	1072.6
Outlet (mg/kg)	69.7	98.9	460.3	488.4	15144.0	20858.1	181.2	271.3	234.8	248.6	1070.5	981.5
Percent difference (%)	-14.59	-16.98	-8.07	29.30	-15.77	-1.87	-6.26	-0.97	-7.74	3.97	-4.09	26.40
Tempe Heights Pond (n = 3 [wet], 4 [dry])												
Inlet (mg/kg)	27.2	37.2	300.8	349.4	13795.5	15323.1	113.7	133.7	104.1	130.2	610.0	697.5
Outlet (mg/kg)	20.2	23.0[b]	146.1	158.0[b]	17440.0	18707.8[b]	204.4	181.2	77.0	85.8[b]	295.9	331.8[b]
Percent difference (%)	30.94	42.55	60.71	55.08	-30.82	-23.54	-13.22	-49.95	28.10	33.60	45.13	55.61
Oakalla Biofiltration System (n = 3 [wet], 3 [dry])												
Inlet (mg/kg)	28.5	36.7	81.1	107.5	18916	23396.5	2932.1	2604.8	52.2	52.7	361.5	599.8
Outlet (mg/kg)	32.7	31.4	89.3	71	16701.3	22836.6	958.5	781.5	30.8	32.9	350.6	226.5
Percent difference (%)	22.1	19.61	22.38	33.97	30.19	17.93	78.69	74.26	50.5	37.57	44.19	62.82

[a] Wet season falls between October and March; dry season falls between April and September.

[b] Significant at the 0.1 probability level (Wilcoxon signed-rank test).

TABLE 8. Labile metal concentrations in water at five stormwater ponds in Greater Vancouver (median values) in wet and dry seasons[a]

	Fe		*Mn*		*Zn*	
	Wet	*Dry*	*Wet*	*Dry*	*Wet*	*Dry*
52Ave/221A Street Detention Facility (n = 4 [wet], 3 [dry])						
Inlet (mg/L)	0.003	0.002	0.028	0.045	0.005	0.003
Outlet (mg/L)	0.007	0.008	0.007[b]	0.152	0.007	0.001
Percent difference (%)	-8.17	-277.14	73.60	-116.92	-38.10	58.30
Griffin Park Biofiltration Pond (n = 5 [wet], 3 [dry])						
Inlet (mg/L)	0.014	0.013	0.005	0.003	0.011	0.007
Outlet (mg/L)	0.003	0.002	0.003[c]	0.011	0.010	0.004
Percent difference (%)	73.12	0.00	35.13	-734.89	20.09	42.11
Westview Interchange Detention Pond (n = 4 [wet], 0 [dry])						
Inlet (mg/L)	0.054	n/a[d]	0.019	n/a	0.031	n/a
Outlet (mg/L)	0.005[b]	n/a	0.009	n/a	0.032	n/a
Percent difference (%)	71.65	n/a	53.60	n/a	12.34	n/a
Tempe Heights Pond (n = 6 [wet], 3 [dry])						
Inlet (mg/L)	0.053	0.012	0.008	0.007	0.011	0.009
Outlet (mg/L)	0.015[c]	0.002	0.014[b]	0.008	0.010	0.002
Percent difference (%)	73.14	81.54	-92.14	-10.26	9.59	66.45
Oakalla Biofiltration System (n = 4 [wet], 2 [dry])						
Inlet (mg/L)	0.017	0.017	0.081	0.069	0.009	0.006
Outlet (mg/L)	0.006[b]	0.003	0.016[b]	0.025	0.004[b]	0.002
Percent difference (%)	71.95	19.95	80.77	62.46	46.11	78.25

[a] Wet season falls between October and March; dry season falls between April and September.
[b] Significant at the 0.1 probability level (Wilcoxon Signed Ranks Test).
[c] Significant at the 0.05 probability level (Wilcoxon Signed Ranks Test).
[d] n/a = not applicable.

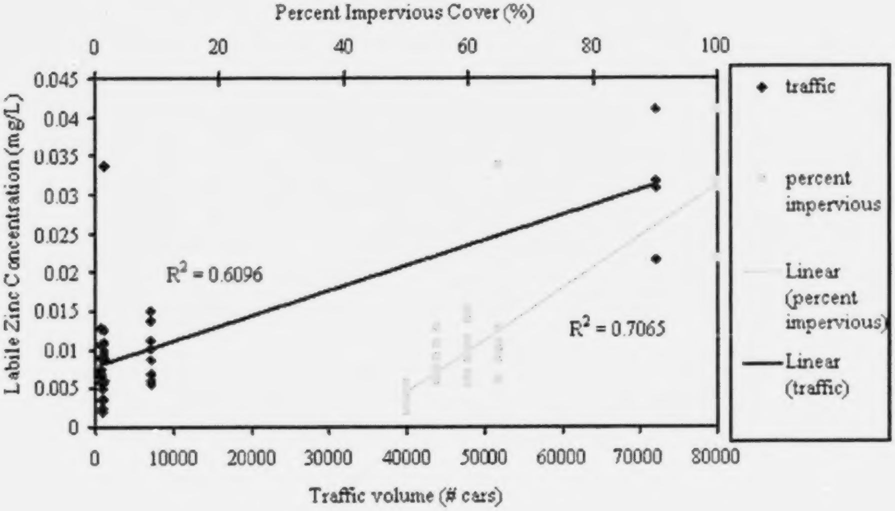

Fig. 10. Relationship between average daily traffic volume and percent impervious cover of the catchments and the labile zinc concentration at the inlets of the five ponds.

pronounced in the wet season (October to April) than during the dry season (May to September). Reduction in Zn concentrations ranged between –4 to +65%, but was only significant in two of the five ponds. Copper, chromium, and lead concentrations in the sediments between the inlet and outlet ranged between –8 to +67% for Cu, –15 to +53% for Cr, and –8 to +72% for Pb. Again, significant differences were only obtained in two of the five ponds. Fe and Mn were generally poorly retained, ranging from –34 to +30% for Fe, and from –38 to +74% for Mn. Only two of the five ponds had positive removal values. These results are similar to those presented in other investigations which also showed the highly variable nature of sediment contamination and detention (Birch et al. 2004; Graney and Eriksen 2004; Pontier et al. 2004).

The labile manganese concentration of the water, measured using the DGT technique, at the inlets was significantly higher at the Oakalla biofiltration system (Table 8). Labile zinc concentrations were significantly higher in the Westview interchange detention pond than all of the other ponds (Wilcoxon signed-rank test, $p < 0.05$).

The labile metal concentrations of iron and manganese were generally lower at the outlet than the inlet, particularly in the wet season, where differences of up to 81% were seen. The difference between inlet and outlet labile zinc concentrations ranged between +10 to +78%, and only one pond had a negative detention (wet season) (Table 8).

An interesting result was found in the consistently positive correlations between the traffic volumes and percent impervious cover of the pond catchments and the concentrations of zinc in the sediment and labile water forms, which suggested that the traffic volumes might be a predictor of zinc accumulation in the ponds (Fig. 10). However, since the study sites had a wide gap in daily traffic volumes, from 960 to 72,000 cars, more intermediary evidence is required to support these observations.

Discussion

The apparent removal of zinc from the water column, as shown by the differences in inlet and outlet concentrations, is likely due to the adsorption of zinc to the clay sediments and subsequent settling to the bottom of the ponds. Iron and manganese were retained poorly in most of the ponds, suggesting effects of mobility from sediments and chemical transformations at the water-sediment interface. The geological nature of these metals contributes to their complex transformations, especially in wetland soils (Goulet and Pick 2001).

The use of DGT proved to be a useful tool for comparing concentrations of labile metals in different areas of the ponds. By measuring the accumulation of many metals over the same time period, a better understanding of the overall condition of the stormwater is obtained. The three-week deployment periods allowed for continuous monitoring of the metals in the ponds, which incorporates the spikes of the first flush often missed by individual grab water samples.

Conclusions

The challenges in measuring and detaining trace metals in sediments in urban watersheds were illustrated in three case studies in the Greater Vancouver area of British Columbia. The results show that a lake in the middle of the Brunette watershed was very effective in reducing the metal contamination downstream, and the analysis of the lake core sediments showed the historic record of metal and selective organic contamination between 1880 and 2003. Metals in suspended sediments transported during storm events showed significantly higher levels of contamination than bed sediments, and in a geochemical extraction it was shown that the Zn and Cu were mostly associated with the organic-sulphur fraction. Toxicity testing based on macroinvertebrate survival and Microtox showed variable results, but comparison of toxicity tests between urban and forested watersheds showed that test organisms had significantly greater mortality and lower growth when exposed to urban sediments.

In a regional comparison it was shown that percent imperviousness was significantly correlated with Co and Ni concentrations in sediments (logarithmic relationships), while Zn and Cu showed a significant linear relationships. At high degrees of urbanization it is evident that the traffic volume is equally as important as percent imperviousness, since two areas with similar percent imperviousness showed significantly different Zn and Cu sediment levels. Again, these two metals are highly reflective of transport intensity, and imperviousness and road density alone do not explain the difference.

In testing the effectiveness of five urban stormwater detention systems, it was shown that the results are highly variable depending on a wide range of physical conditions, land use activities, traffic volumes, and detention system designs. The range of total metal retention was between –15 to +72% for Cu, Pb, and Zn, while Fe and Mn retention was generally poor. The labile Zn concentrations were more effectively retained in four of the five ponds. The most interesting results were found between percent impervious cover, traffic volume, and the Zn concentration in sediments and labile form.

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A New Approach in Measuring Rainfall Interception by Urban Trees in Coastal British Columbia

Yeganeh Asadian[1]* and Markus Weiler[2]

[1] *Department of Forest Resources Management, The University of British Columbia, Vancouver, BC V6T 1Z4, Canada*
[2] *Institute of Hydrology, The University of Freiburg, Fahnenbergplatz 79098 Freiburg in Breisgau, Germany*

Interception loss plays an important role in controlling the water balance of a watershed, especially where urban development has taken place. The aim of this study was to illustrate the importance of urban trees as a form of 'green infrastructure' where they reduce stormwater runoff and rainwater intensity. In addition, trees cause a delay in precipitation reaching the ground. Interception loss was studied in the North Shore of British Columbia. We applied a unique methodology for measuring throughfall under six different urban trees using a system of long polyvinyl chloride pipes hung beneath the canopy capturing the throughfall and draining it to a rain gauge attached to a data logger. Different tree species (Douglas-fir [*Pseudotsuga menziesii*] and western red cedar [*Thuja plicata*]) in variable landscape sites (streets, parks, and natural forested areas) and elevations were selected to ensure that the system adequately captured the throughfall variability. Interception and throughfall were monitored over a one year cycle for which the results of seven discrete storm events for coniferous trees from the District of North Vancouver during 2007 to 2008 are presented. Cumulative gross precipitation for seven selected events was 377 mm. Average canopy interception during these events for Douglas-fir and western red cedar were 49.1 and 60.9%, where it corresponded to average net loss of 20.4 and 32.3 mm, respectively. The interception loss varied depending on canopy structure, climatic conditions, and rainfall characteristics.

***Key words:* urban environment, throughfall, interception loss, stormwater runoff**

Introduction

Urbanization has resulted in profound changes to natural watershed conditions by altering terrain, vegetation, soil characteristics, and surface conditions. Urban development impacts climatic conditions and alters the hydrological processes leading to more flashy runoff and increased pollution in urban watersheds (Sanders 1986; McPherson et al. 1997). The losses in vegetation cover and increases in impervious surfaces, such as paved roads, sidewalks, and concrete buildings, increase the total amount of runoff, the flashiness of runoff events, flooding, erosion, and the cost of stormwater management. Villarreal and Bengtsson (2004) noted that stormwater runoff prior to development was regulated by trees, vegetation, and natural soils. Trees and soil function together to reduce stormwater runoff. Trees reduce stormwater runoff by intercepting rainwater on leaves, branches, and trunks. Some of the intercepted water evaporates into the atmosphere and some infiltrates into the ground, decreasing peak flows and the total amount of urban runoff. Trees also slow storm flow events by reducing the volume of water that must be managed at one time and the rainfall intensity. Trees are generally overlooked in urban planning, but they are an integral component of the urban infrastructure, capable of controlling the hydrological processes, regulating air and water quality, reducing Urban Heat Islands (UHI) and absorbing CO_2 (Sanders 1986; Taha 1997).

Stormwater managers have started to use trees as a tool to help reduce stormwater generation and, in this way, reduce the cost of constructing traditional stormwater control infrastructure. The value of the tree for stormwater management has been calculated based on the avoided costs of handling stormwater runoff (McPherson et al. 1997; Zipperer et al. 1997; Villarreal and Bengtsson 2004). McPherson et al. (2005) reported that in some cities in the U.S.A., the urban tree investment can be between $13 to $65 per tree annually in planting and maintenance cost. In return, gains in stormwater services are between $1.37 to $3.09 per dollar that would have otherwise been invested in traditional stormwater management. Another study has estimated the worth of the U.S.A.'s urban forests as $400 billion in terms of stormwater management mitigation alone (American Forests 1996). These studies demonstrate the importance of trees as source controls capable of treating stormwater at the site level by reducing the runoff component within the hydrological cycle.

Urban vegetative cover is arranged as individual or stands of trees that contribute to the sustainability of the environment. From an urban hydrological point of view, the most noticeable effect of vegetation is rainfall interception by the canopy (Xiao and McPherson 2002; Guevara-Escobar et al. 2007; McJannet et al. 2007a, 2007b). Canopy interception losses frequently modify the intensity and distribution of precipitation reaching the ground. Trees retain water on site temporarily or permanently, slowing the flow to waterways. Horton (1919) and Rutter et al. (1975) classified canopy interception (I_{net}) into various components: a fraction of gross precipitation (P_G) that falls as throughfall (p), the proportion that is diverted to stemflow (p_t), and the proportion that is stored and evaporated. Canopy interception represents the difference between gross precipitation (above canopy) and net precipitation (below canopy) (Jetten 1996; Aboal et al. 1999; Xiao et al. 2000a, 2000b).

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* Corresponding author: yeganeh@interchange.ubc.ca

Previous studies on rainfall interception, primarily carried out in naturally forested areas, report a wide range of values for interception losses, throughfall, and stemflow. Interception loss is commonly 20 to 40% in coniferous, and between 10 to 20% in deciduous forests (Crockford and Richardson 1990; Llorens et al. 1997; Link et al. 2004; Llorens and Domingo 2007). The amount of interception loss is highly dependent on forest structure (e.g., species, dimension, density), canopy structure (e.g., foliation period, leaf and stem surface areas, gap fraction, surface detention storage capacity) and meteorological factors (e.g., rainfall amount, duration, intensity, frequency, temperature, wind, humidity) (Crockford and Richardson 2000; Xiao and McPherson 2002; Nadkarni and Sumera 2004).

While considerable research concerning the impact of tree interception loss on hydrological processes has been conducted in forested areas, the effects of urban trees on rainfall interception and runoff have not been well quantified. The characteristics of trees in forested areas are different from those in urban settings in terms of available growing space, canopy cover, age, diversity, and microclimate (Zipperer et al. 1997; Xiao and McPherson 2002; Wang et al. 2008). The measurement and monitoring methods involved in closed forested areas ranged from troughs and rain gauges to plastic sheets; however, it has been suggested that most of these common sampling techniques cause large errors in the estimated interception (Horton 1919; Xiao et al. 2000b; Link et al. 2004). It is important to note that sampling design that captures the throughfall variability is key in determining the accuracy and time resolution of obtained data (Lundberg et al. 1997; Keim et al. 2005). These studies illustrate that rainfall interception by forests is extremely variable and difficult to measure.

In urban settings, field observations and experimental measurements of rainfall interception processes are sorely needed in order to better understand these processes. Urban tree interception processes are somewhat different from those reported for natural forests as a result of various factors such as edge effect, isolation (greater distances between individuals), open canopies, higher temperatures, and wind penetration and associated rainfall (Zipperer et al. 1997; Guevara-Escobar et al. 2007). These characteristics define the storage capacity for each stand or individual tree, and control the evapotranspiration rate.

The objective of this paper was to gain a better understanding of the rainfall interception processes for a large number of urban trees, and to quantify the throughfall and estimate the interception losses using an innovative monitoring approach.

Methods and Materials

To address the objectives of this research with regard to rainfall interception, throughfall for six coniferous trees was measured along the North Shore (North Vancouver) in British Columbia during 2007 to 2008. The selected trees were located on private and public properties along streets, parks, and in forested areas.

Study Site and Climate

North and West Vancouver are highly urbanized cities with increasing urban development that resulted in the creation of larger proportion of impervious surfaces. The dominant land use in these municipalities is residential, followed by industrial and commercial areas. The major rivers and creeks pertaining to these areas are: Capilano, MacKay, Mosquito, Lynn, and Seymour. The major concern regarding these waterways is the direct drainage of stormwater runoff into the rivers leading to flooding and nonpoint sources of pollution (Environment Canada 2007).

The regional climate is characterized by cool, wet winters and warm, moderate summers. In Vancouver it is common to have more than 166 days per year with measurable precipitation on average. These coastal rainfall events are described as long durations with low intensities. The average annual precipitation near sea level ranges from 1,200 to 3,000 mm at higher elevations, with most of the rainfall occurring in the winter. The amount of precipitation varies with elevation, increasing by about 100 mm for every 100 m rise in altitude. Consequently, the North Shore receives more rain and snowfall at higher elevations during the winter. The average annual temperature is 10°C at sea level (Environment Canada 2007).

Experimental Design and Instrumentation

Tree selection. The experimental setup focused on the direct measurement of throughfall for coniferous trees. The main coniferous species selected were Douglas-fir (*Pseudotsuga menziesii*) and western red cedar (*Thuja plicata*). The trees were classified into different types: dominant, codominant, single, and forested area (control). Dominant species were the main overstory trees in a plant community, which contributed the most cover or basal area to the community. Trees with crowns receiving full light from above, but comparatively little from the sides, were defined as codominant species. Single standing trees were exposed to light and wind from all sides. Forested areas were used as control sites, where trees were embedded within large groups of trees independent from any edge effect; these areas have little or no development (Oke et al. 1989; Zipperer et al. 1997; Brooks et al. 2003).

Tree health condition was also assessed because it reflects the structural integrity. This assessment helped indicate patterns of throughfall for individual trees. The rating of tree condition involved analysis of the tree crown and the density of foliage. Two different classes were assessed based on density of the canopies for coniferous trees as good and poor. Tree health conditions in control sites were not evaluated as they were considered to be away from urban areas. These controls were assumed to be representative of health conditions in forested settings, which are naturally variable.

Throughfall measurement. The experimental unit built under each tree consisted of four components: a wooden frame, PVC (polyvinyl chloride) pipes, a rain gauge, and a data logger. The wooden structure included a platform on which the rain gauge was placed. Four metal rods supported a wooden roof and held the platform together. This frame was mounted directly to the trunk of the tree. Two PVC pipes were used per experimental unit. They were hung at an angle from branches using ropes and bolts. The two pipes were positioned underneath the canopy of each tree based on the shape and structure of the tree in a way that the entire diameter of the canopy was covered. The length of each pipe was approximately 3-m long, where three 0.85-m by 0.028-m slits were cut on top along the length of each pipe providing the total surface area of 0.1428 m^2. The throughfall was captured by these openings and drained into a tipping bucket rain gauge (RAINEW, RainWise Inc., Bar Harbor, Maine). Data loggers (HOBO, Onset Computer Corporation, Pocasset, Mass.) attached to the rain gauges recorded both the air temperature and rainfall events of the canopy. The temperatures recorded by the data loggers accounted for within-canopy temperature variation, which is suggested to change along the vertical gradient and have a minor impact on canopy interception responses (Jetten 1996; Brooks et al. 2003). Overall, this flexible system allowed independent movement of the different components of each experimental unit without causing any serious damage to the entire structure (Fig. 1).

Meteorological station. Gross precipitation was measured using control units of the same design (Fig. 2). These units were positioned on the rooftops of buildings away from any structures that may block rainfall. Figure 3 illustrates the proximity of the six study sites in the District of North Vancouver to the rain gauge on the rooftop. Additional climate stations were set up in each municipality to capture the meteorological variability along the elevation gradient. These climate stations were within a 5-km radius of the study sites. Each station was equipped to measure barometric pressure, temperature, humidity, rainfall, wind speed, and direction. These supplemental records were utilized to validate the tipping rain gauge data, thus ensuring correct identification of rainfall events.

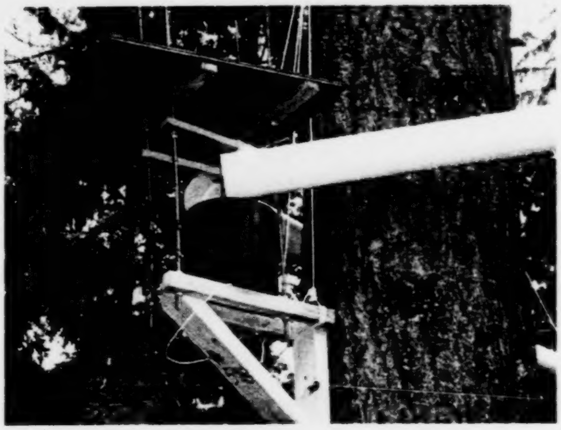


Fig. 1. The rainfall interception measuring system.

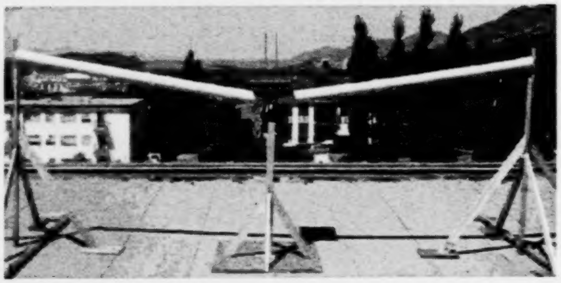
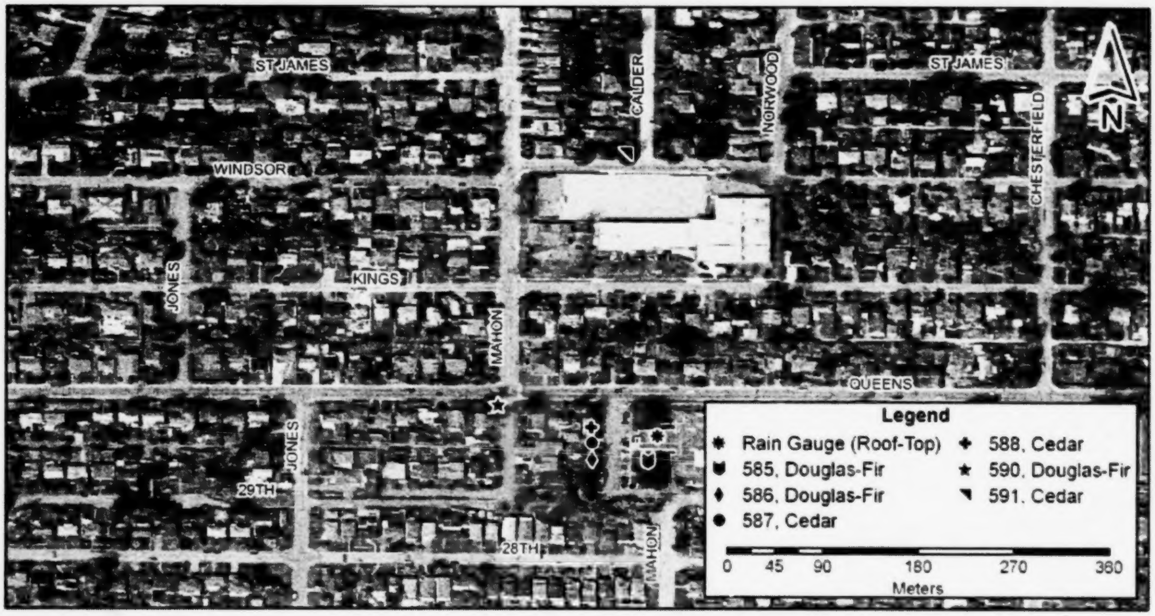


Fig. 2. Reference experimental system installed on the rooftop of North Vancouver's City Hall to measure gross precipitation (above canopy rainfall).

Data collection and calibration. The data loggers were programmed to record the number of tippings where these numbers were converted into total amount of gross precipitation/throughfall and intensity readings. The climate stations and rain gauges were calibrated after the installation in the field.

Methods for calculation. Table 1 shows the location and assessed attributes for the selected trees. Rainfall events were defined as storms with cumulative gross precipitation exceeding 1 mm, with a minimum of four hours without precipitation between events. Numerous events were eliminated due to clogging of rain gauges by leaves in late autumn and ice during the winter season when temperature fell below 0°C.

In this study, stemflow was not measured since it is considered to be a minor component of the water balance for mature canopies especially conifers, where the branches slope downward from the stem. This structural characteristic minimizes the probability for intercepted water to be routed to the stem, even if a small amount of precipitation intercepted in the upper canopy


ST JAMES
ST JAMES
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WINDSOR
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JONES
MAHON
QUEENS
JONES
29TH
28TH
MAHON
Legend
Rain Gauge (Roof-Top)
585, Douglas-Fir
586, Douglas-Fir
587, Cedar
588, Cedar
590, Douglas-Fir
591, Cedar
0
45
90
180
270
360
Meters

Fig. 3. Locations of study sites and the rooftop rain gauge in the District of North Vancouver (orthophotos provided by District of North Vancouver's GIS Department).

TABLE 1. Characteristics of the selected experimental trees including their location and tag numbers

Tag #	*Tree species*	*Longitude*	*Latitude*	*Tree type*	*Condition*	*Height (m)*	*Diameter at breast height (cm)*
586	Douglas-fir	-123.078936	49.336042	Dominant	Poor	32	66
585	Douglas-fir	-123.078469	49.335931	Single	Good	35	78
590	Douglas-fir	-123.079685	49.336425	Single	Poor	39	68
587	Cedar	-123.078917	49.336200	Codominant	Good	29	70
588	Cedar	-123.078894	49.336311	Codominant	Poor	29	43
591	Cedar	-123.078558	49.338600	Single	Good	32	95

TABLE 2. Rainfall depth from two other nearby stations[a]

Event	Duration of Measurement	*Rain gauge on*[b] *therooftop of District of North Vancouver*	*Standard climate*[b] *station in the District of North Vancouver*	*Rain gauge on*[c] *rooftop of City of North Vancouver*
		P_G (mm)	*P_G (mm)*	*P_G (mm)*
1	October 15–23, 2007	209.6	205	205.9
2	November 3–4, 2007	26.6	31.2	28.6
3	December 18–19, 2007	39.7	39.6	37.8
4	January 2–3, 2008	28.2	31.8	29.9
5	March 16–17, 2008	22.6	25.2	20.6
6	May 13–14, 2008	26.3	24	—[d]
7	June 9, 2008	23.8	24.2	—

[a] P_G = gross precipitation.

[b] Elevation = 130 m.

[c] Elevation = 110 m.

[d] Missing data for two events.

still contributes to stemflow. In addition, the bark is ridged and ruffled where it absorbs greater amounts of water. The absorption of water by epiphytes and various moss species on branches and tree trunks also play a role in controlling the stemflow. As a result we assumed the stemflow to be insignificant, based on the results of previous research studies (Crockford and Richardson 1990; Brooks et al. 2003; Link et al. 2004; Llorens and Domingo 2007).

For this investigation we computed the total volume of throughfall captured underneath each tree for individual events by using the total number of tippings and the obtained volume from calibration. To note, the difference between the surface areas of the PVC pipes and the rain gauge (which is 4.2) was taken into account. Canopy interception was derived from the difference between the P_G and p for individual events. The data for P_G was obtained from the reference climate station on the rooftop of the District of North Vancouver; however, for comparison, Table 2 includes P_G from a standard climate station in the District of North Vancouver, and a nonstandard rain gauge on the rooftop of North Vancouver's City Hall.

Results

The high spatial and temporal resolution of the throughfall data enabled us to determine canopy interception losses for a wide range of trees (Xiao et al. 2000b; Link et al. 2004). The selected trees were differentiated by species, type, and health condition.

Seven discrete storm events were chosen between October 2007 and June 2008. Table 3 highlights the event characteristics. Selected events generated 377 mm of gross precipitation with a maximum hourly rainfall intensity of 13.3 mm/hr. This intensity corresponds to a two-year event in this area (Denault et al. 2006). These obtained results reflect on the rainfall characteristics in the North Shore, where the frontal system during October through April produces long durations and relative low rainfall intensities.

Climate and Precipitation Variability During an Event

Precipitation and above- and within-canopy climate data for Douglas-fir and western red cedar for event 3 are shown in Fig. 4 and 5. These figures show the effect of the urban tree canopies on throughfall intensity, and demonstrate the range of conditions controlling interception loss during the rainfall event.

Event 3 began at 0500 hours on December 18 and lasted 39 hours. During this period, 39.7 mm of precipitation was recorded by the reference rain gauge on the rooftop of North Vancouver's District Hall. Precipitation intensity, humidity, wind speed, and temperature were typified as moderately low. Figures 4a and 5a illustrate that there was not much variation between the measured temperatures above and below the canopy for both species. Wind speed was recorded below 0.1 $m{\cdot}s^{-1}$, indicating absence of wind during the event. Average humidity was above 95%. The amount of throughfall captured underneath each canopy averaged 50.1 and 46.2% (19.9 mm and 18.3 mm) for Douglas-fir and western red cedar, respectively.

Figures 4b and 5b show that throughfall levels for both species are not constant, but they are dynamic. The difference in gross precipitation and net precipitation magnitude is shown in Fig. 4c and 5c. Table 4 presents the delay in throughfall reaching the ground for all study sites. The delay ranged from 6 to 7.5 hours for event 3. This delay did not affect the peak in net precipitation; however, as shown by Xiao and McPherson (2002), this delayed the peak runoff for a storm. Throughfall ceased roughly 3.8 hours after the rainfall stopped.

The average rainfall intensity for event 3 was determined by dividing the gross precipitation by the rainfall duration. Figures 4c and 5c illustrate the impact of canopy on throughfall intensity, and exemplify how the climatic conditions control evaporation during the rainfall event. Both temperature and wind are suggested to play an important role in driving the evaporation rate, however, wind was omitted due to low velocities, which were less than 0.1 $m{\cdot}s^{-1}$ (Brooks et al. 2003; Link et al. 2004).

TABLE 3. Event characteristics

Event	*Duration of Measurement*	*Duration (h)*	*P_G (mm)*	*Average rainfall intensity (mm/h)*	*Maximum Rainfall intensity (mm/h)*	*Average temperature (°C)*	*Wind speed (ms^{-1})*
1	October 15–23, 2007	179	209.6	1.1	13.3	9.6	0.006
2	November 3–4, 2007	31.2	26.6	0.8	7.3	8.6	0.03
3	December 18–19, 2007	39	39.7	1.0	6.9	4.8	0.03
4	January 2–3, 2008	24	28.2	1.2	8.7	6.1	0.03
5	March 16–17, 2008	20	22.6	1.1	3.3	3.5	0.006
6	May 13–14, 2008	25	26.3	0.9	3.0	9.7	0.006
7	June 9, 2008	15	23.8	1.6	8.4	8.6	0.006

[a] P_G = gross precipitation.

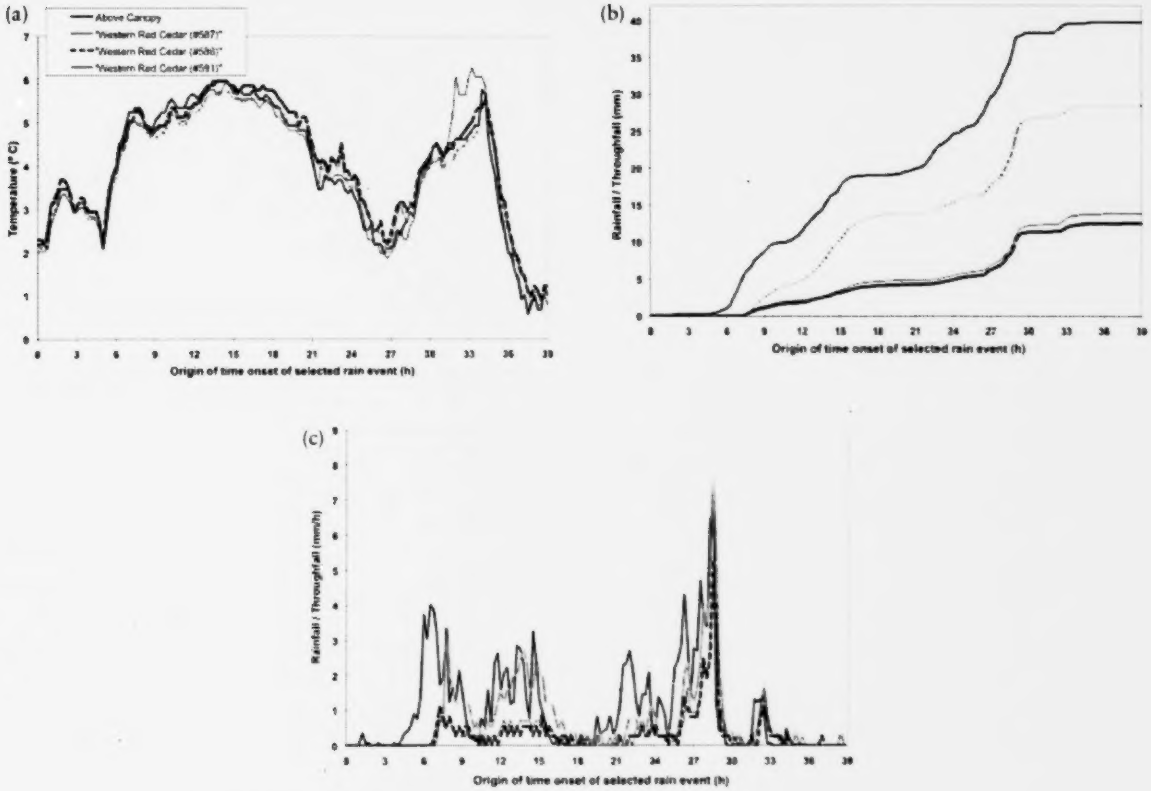

Fig. 4. Meteorological and throughfall data for rainfall event three (western red cedar).

TABLE 4. Delay in throughfall with respect to gross precipitation

Tag #	*Event 1 Start time (h)*	*Event 2 Start time (h)*	*Event 3 Start time (h)*	*Event 4 Start time (h)*	*Event 5 Start time (h)*	*Event 6 Start time (h)*	*Event 7 Start time (h)*
586	28	0.2	9.5	2.5	0.2	11.0	4.2
585	—	—	7.5	2.5	0.2	8.7	4.7
590	8.5	0.2	6.5	2.2	0.2	0.2	1.0
587	27.7	0.2	7.5	2.7	0.5	6.7	4.0
588	28	0.2	7.2	3.0	0.5	10.5	2.0
591	45.5	1.0	6.5	2.7	0.2	6.5	5.2

TABLE 5. Precipitation and interception summary[a]

Tag #	*Event 1 I_{net} (%)*	*Event 2 I_{net} (%)*	*Event 3 I_{net} (%)*	*Event 4 I_{net} (%)*	*Event 5 I_{net} (%)*	*Event 6 I_{net} (%)*	*Event 7 I_{net} (%)*
586	44.4	24.0	49.5	9.0	70.5	98.3	78.3
585	—	—	52.6	25.1	76.9	88.5	77.3
590	10.9	15.4	48.10	17.4	57.9	80.5	60.3
587	40.3	31.3	28.3	5.4	53.1	70.8	60.1
588	68.6	67.5	68.4	49.2	76.0	95.3	72.8
591	73.7	56.6	65.20	51.2	70.6	94.0	80.3

[a] I_{net} = canopy interception.

Throughfall and interception loss. Interception loss was defined as the difference between the total gross and total net precipitation. Table 5 summarizes throughfall and interception losses for the selected trees and events. For events 1 and 2 there are no data available for one of the Douglas-fir trees (# 585) due to rain gauge failure (clogging).

When evaluating the average interception losses, it is evident that two of the cedar trees (# 588 and 591) showed the highest interception losses compared with the other selected trees during the events in the fall and winter. Both western red cedar trees were codominant; however, one was of good health condition and the other, poor. Events 5, 6, and 7 occurred during the spring and summer, where both Douglas-fir and western red cedar had high interception losses. The highest interception loss calculated was in event 6 by a dominant Douglas-fir of a poor condition. Based on the results, compared with western red cedar, the Douglas-fir trees showed a wider range of interception losses during the seasons.

Event 5 was the smallest precipitation event with relatively moderate levels of interception loss for all selected trees. Event 7 had the shortest duration in comparison with the other selected events. The highest interception loss was seen for event 6 with 26.3 mm of gross precipitation over 25 hours. The average temperature was recorded as 9.7°C with a maximum rainfall intensity of 3.0 mm/hr. In general, rainfall type plays a role in determining interception loss. For instance, a low intensity, long-duration frontal rainfall generates a different interception loss than a high intensity short duration convectional storm (Xiao et al. 2000a, 2000b; Pypker et al. 2005; Deguchi et al. 2006).

Discussion

The measurement of throughfall using gauges under individual tree canopies has been successfully conducted to estimate interception loss. The methodology applied in this research is innovative and has not been applied in any other research within urban environments. The measuring system was easy to build and install, and the design minimized evaporation and splashing. These experimental units had the ability to collect spatially variable throughfall underneath the canopy.

Net interception loss is determined as the difference between gross precipitation and the sum of throughfall and stemflow. In this study, we did not include the stemflow since it was assumed to be negligible. Based on the obtained throughfall data, the average percent I_{net} for

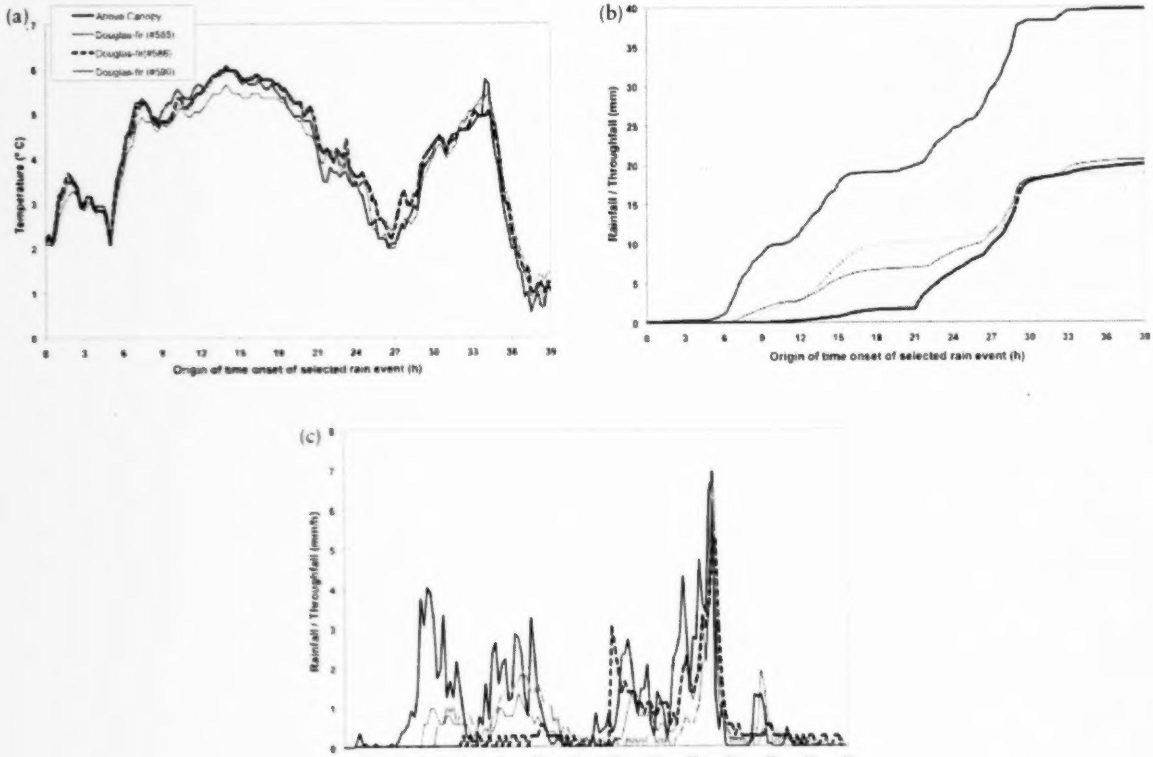

Fig. 5. Meteorological and throughfall data for rainfall event three (Douglas-fir).

the seven events ranged between 17 and 89%, which were 4.8 and 22.0 mm of gross precipitation respectively. The lowest interception losses occurred during event 1 and 4. Based on the variability in rainfall amount, intensity, and duration the interception losses for coniferous trees in this study ranged from 5 to 98% (1.5 to 24.3 mm with reference to the amount of gross precipitation).

Our results suggest that interception losses for coniferous trees are significantly higher within urban environments compared with forested areas. Link et al. (2004) suggested that in temperate forests, annual interception losses for coniferous canopies ranged from 9 to 48%, while Bryant et al. (2005) reported 22.3% interception loss in a pine forest. The interception values obtained in our study suggest that the interception losses for trees in urban environments are twice as much as trees in natural forested areas. Possible factors contributing to these differences are UHIs (urban areas which have significantly higher temperatures than the surroundings), greater distances between trees (edge effect), and open grown canopies.

UHIs cause local-scale variation in temperature differences between urban and natural forested areas. Taha (1997) stated that the temperatures within urban areas tend to be higher due to replacement of natural vegetation by man made structures, consequently resulting in less evapotranspiration. In addition, urban trees are isolated with greater distances between them, making them more exposed during severe weather events, unlike trees within forested areas where they are surrounded by other trees (Aboal et al. 1999; Nadkarni and Sumera 2004). High wind during a rainfall event can mechanically shake precipitation from the canopy and thus reduce interception loss. Winds during evaporation can either shake precipitation loose, or increase the rate of evaporation and decrease the time until maximum interception capacity is attained. Urban tree canopies are classified as open grown trees as a result of no intertree competition; consequently, they have larger structural dimensions (e.g., larger storage capacity) than trees in forests (Horton 1919; Zipperer et al. 1997; Brooks et al. 2003). In our study, UHIs, isolation, and open canopies attributed higher interception losses by urban trees.

Tree health condition and type were also found to affect interception rate. Single standing trees in good health were expected to have a higher interception rate. This was demonstrated in some, but not in all the events. A codominant western red cedar of poor health intercepted at higher rates compared with other trees of better health conditions and types. Also, a dominant Douglas-fir with poor canopy condition showed the highest interception rate for event 6. Western red cedar trees generally had higher interception losses compared with Douglas-firs. This is due to the differences in canopy structure between the two tree species. For the events between March and June, the interception losses were relatively high for both species. The high rates can be explained by small rainfall events, where most of the water from the event is used to wet the crown surfaces.

The time delay in throughfall penetrating through canopy was greatest for event 1 (8.5 to 45.5 h) in comparison with events 2, 4, and 5 where there were no significant delays. Events 3, 6, and 7 had moderately higher time delays, however, lower than event 1. Tree type and health condition played an important role in controlling the time delay. For the seven events it was noticed that the dominant and codominant trees with good health conditions showed a longer time delay in throughfall. Also, when comparing the time delays between the two species, western red cedar showed later delays than Douglas-fir trees.

Trees generally dampen rainfall intensity; however, there were instances where the throughfall intensity was equivalent or higher than the actual rainfall intensity. The highest throughfall intensity was seen in event 3 at 28.7 h. The throughfall intensity exceeded the actual rainfall intensity by single standing Douglas-fir and a codominant western red cedar. The calculated rainfall intensity was 6.0 mm/h while the througfall intensities for Douglas-fir and western red cedar were recorded at 7.3 and 8.0 mm/h. This variation can be explained by rainfall characteristics, meteorological factors, and structure of the canopy. It is evident in Fig. 4c and 5c that high throughfall intensity is delayed in time for lower rainfall intensities. Crown density wetness is another factor to consider. As the crown dampens, the drip becomes larger, consequently resulting in higher throughfall intensities. Another reason can be suggested to reflect on crown wetness. As the crown dampens the drip becomes larger, consequently resulting in higher throughfall intensities (Crockford and Richardson 2000; Brooks et al. 2003).

The observed reduction in throughfall intensity by tree canopies serves two purposes. First, it delays water reaching the ground by temporary storage of the water on the tree. This storage both reduces and delays the peak in the stormwater runoff. Second, it protects the mineral soil surface from the energy of raindrops reaching the ground at maximum velocity. Reduction of raindrop energy by interception minimizes soil detachment and subsequent erosion, which in turn protects soil structure and infiltration capacity leading to less stormwater runoff (Xiao and McPherson 2002; Pypker et al. 2005). All the selected events demonstrated reduction in raindrop energy by having lower intensities captured underneath the canopy. The differences in the magnitudes of rainfall intensity for the events were dependent on climatic conditions. A tree's health condition, type, and species can be suggested to contribute to the differences in throughfall intensities.

Our results confirm that canopy interception loss is greater in urban areas due to isolation, open canopies, and higher temperatures. These higher interception losses play an important role in controlling stormwater runoff. This novel perspective of the interception processes in nonforested settings may prove to be useful for modelling future impacts of large-scale urban tree plantings on interception and runoff.

Conclusion

This study evaluated the interception losses for coniferous species in North Vancouver. Interception losses calculated for urban trees were approximately twice as great as those calculated for trees within natural forest stands. The identified controls on interception loss were meteorological factors, tree type, and health. The results were variable depending on location, tree health, and canopy structure. The interspecies variation on interception was evident as western red cedar trees showed higher interception losses, longer time delays, and lower throughfall intensities compared with Douglas-firs.

The goal of this project was to shed some light on rainfall interception by single and stands of trees in urban environments in order to provide data, models, and additional information for planners, developers, and municipal engineers to utilize in the planning of future urban development. Using natural vegetation as a low impact development and best management practice is an effective technique as it controls stormwater runoff on site, mitigating the impacts of urbanization on urban hydrology at a local scale (Graham et al. 2004).

Acknowledgments

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Treatment Performance of an Extensive Vegetated Roof in Waterloo, Ontario

Kyle Vander Linden[1] and Micheal Stone[2]*

[1] *School of Planning, University of Waterloo, Waterloo, ON N2L 5G1*
[2] *School of Planning and Department of Geography, University of Waterloo, Waterloo, ON N2L 5G1*

Vegetated roof technologies are used as treatment measures to mitigate the effects of urban stormwater. A mass balance approach was used to assess the treatment performance of a vegetated roof located on the City Hall in Waterloo, Ontario. The vegetated and control roof sections were instrumented to measure precipitation, storage, and outflow for 18 storm events from June to October, 2006. Concentrations of suspended solids, total phosphorus (TP), and SRP (SRP) in precipitation and roof (vegetated and control) runoff were measured. A total of 155.6 mm of rain fell during the study period. The vegetated roof retained 64.5 mm (43.9%) of the total rainfall while the control roof retained approximately 5.1 mm (4.0%). For individual rain events, the vegetated roof retained an average of 3.5 mm (47.6%), while the control roof retained approximately 0.3 mm (4.7%). Water retention varied with storm size, season, and frequency of storm events. The vegetated roof retained 80.1% of precipitation for storm events ≤3.5 mm, and 34.9% for storm events >3.5 mm. TP and SRP concentrations from the vegetated roof were significantly higher than either the precipitation or runoff from the control roof.

***Key words*: green roof, source control, treatment performance, water quality, suspended solids, phosphorus**

Introduction

The effects of stormwater runoff on the health of aquatic ecosystems (Novotony and Olem 1994), flooding (U.S. EPA 1999; Paul and Meyer 2001), and drinking water supplies (Marsalek et al. 2006) are well documented. Accordingly, there is increasing interest globally to implement sustainable urban drainage systems (SUDS) which employ a variety of source, conveyance, and end of pipe controls designed to reduce the volume of stormwater runoff (Marsalek and Chocat 2002; Graham et al. 2004). Vegetated roofs are increasingly being recognized as one source control measure that could be effectively implemented in both new and retrofit buildings to reduce runoff volume (TRCA 2006). Much of the innovation in vegetated roof technology, policy, and research (Ngan 2004; GRHC 2005; Cannata 2005; Getter and Rowe 2006) was initially developed in Germany and has subsequently been used to develop regulations in Europe that promote the application of vegetated roofs for stormwater management (Ngan 2004; GRHC 2006). In North America, however, several barriers (financial constraints, public awareness, quantifiable research, technical expertise, and industry standardization) limit broad application of vegetated roof technology as a viable source control measure (Getter and Rowe 2006).

The application and treatment performance of various vegetated roof technologies vary due to a range of climatic conditions which limit broader adoption of the technology (VanWoert et al. 2005). The implementation and effectiveness of vegetated roofs can vary at the city and regional scale and especially between localities that have financial incentives for vegetated roof construction, i.e., Toronto, Chicago, and Portland (Peck and Goucher 2005; Getter and Rowe 2006). Consequently, there is a need to assess the treatment performance of vegetated roofs across varying geographical locations, and to develop and implement relevant policies for their use as source control measures. Such information is required to create vegetated roof building and performance standards, and to lower cost by adopting financial incentives or stormwater fee rebates that may encourage green roof industry growth.

Given the need to increase knowledge of the utility of vegetated roofs as a viable source control measure, the objective of this paper was to evaluate the treatment performance of an extensive vegetated roof located on a municipal building in the City of Waterloo, Ontario. The vegetated roof and a control roof were instrumented to measure precipitation inputs, storage, and runoff during 18 storm events that occurred between June 2 and October 22, 2006. The rates and magnitude of water storage and loss from the vegetated roof and a control roof are reported. Event mean concentrations and temporal variability of suspended solids (SS), total phosphorus (TP) and SRP (SRP) in precipitation and roof runoff are presented.

Methods

Description of Study Site

The vegetated and control roof are located on the Waterloo City Hall building in Waterloo, Ontario

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* Corresponding author: mstone@uwaterloo.ca

(43°28'02.16"N, 80°30'59.44"W). The vegetated roof was installed September, 2005 and covers an area of 1,650 m^2. The section of vegetated roof monitored in this study has an area of 424.3 m^2, defined by the drainage bay. Structurally, the vegetated roof consists of vegetation and growth medium, water retention fleece, a drainage layer, and a root resistant waterproof membrane. The vegetation layer is composed of a *XF 301* precultivated sedum-moss combination blanket composed of eight species of *Sedum spp*. and nylon mesh filled with a growth substrate. The growth substrate was a *XF xero terr* growing mix and consisted of a 35-mm mineral substrate composed of 60% porous materials (inert crushed brick, pumice, or expanded slate). The growing mix had a maximum particle size of 1 mm and was comprised of 25% fine washed sand, 14% weed free organic compost, and 1% dolomite (Xero Flor 2006a). A 12-mm *XF 158 D* water retention fleece (composed of synthetic fibres of polyester, polyamide, polypropylene, and acrylic) with a water holding capacity of 1,200 g/m^2 was located beneath the growth substrate (Xero Flor 2006b). An *XF 108 H* drainage filter fleece with water holding capacity of 800 g/m^2 of water was used to filter excess draining water. The bottom layer of the vegetated roof was a *XF 112* root resistant water membrane composed of a polyethylene sheet that prevented root and water penetration (Xero Flor 2006b). The vegetated roof was 6.2-cm thick, weighed 45.9 kg/m^2, and held 28.8 L/m^2 when fully saturated. An irrigation system was not installed with the vegetated roof; rather, manual watering was included as part of the vegetated roof maintenance. The control roof (no vegetation) was covered with a bituminous single ply material, located approximately 30 m from the vegetated roof and drained an area of 246.61 m^2.

Meteorological Data

A meteorological station was located on the roof of the Waterloo City Hall. Wind speed was measured with a Wind Speed Smart Sensor (± 1.1 m/sec [2.4 mph]). Ambient air temperature and relative humidity were measured with a Temperature/RH Smart Sensor. Precipitation was measured with a tipping bucket rain gauge (±1.0 % at up to 20 mm/hour). Data from all sensors were stored on an Onset Computer Data Logger at 5 minute intervals and downloaded with a USB cable to a laptop computer using HOBO weather station software.

Flow Measurement and Sample Collection

To measure runoff, cylinder weirs were inserted into the drains of both the vegetated and control roofs and sealed with a marine sealant. The weirs were calibrated for a range of flow rates using a garden hose, and a corresponding weir height (cm) was measured with a 730 Bubble Module (±0.015 m) (ISCO 2003) and recorded by the ISCO 6700. These data were used to program the ISCO 6700 water sampler with Flow link software to convert weir height to discharge.

Composite rainfall samples were collected in an acid-washed triple-rinsed 18-L bottle connected to a tipping bucket rain gauge. Samples of runoff from the vegetated and control roofs were collected with an ISCO 6700 automatic sampler. A two-part, flow-weighted program (Part A and B) was used to sample runoff. During Part A, six duplicate runoff samples were collected every 28 L, and during Part B, six duplicate samples were taken every 280 L.

The treatment performance of the vegetated roof was determined by measuring percent effluent flow reduction and percent concentration reduction using the following two equations:

$$\%\text{ effluent flow reduction} = 100(Q_i - Q_o)/Q_i \qquad (1)$$

$$\%\text{ concentration reduction} = 100(C_i - C_o)/C_i \qquad (2)$$

where Q_i = rainfall depth (mm), Q_o = runoff (mm), C_i = contaminant concentration, C_o = effluent contaminant concentration (µg/L; mg/L), and A = Area (m^2) (Mulamoottil et al. 1999).

Water Quality

The temperature, conductivity, and pH of each sample was measured in the laboratory using an Orion 105A+ Conductivity Meter and Orion Conductivity Cell (± 2%) following Standard Method 2510 B (APHA et al. 1995). The pH of each sample was measured with a calibrated Orion 250A and Orion pH probe (±2%) according to Standard Method 4500 H$^+$ (APHA et al. 1995). SS concentrations were determined by filtration (Standard Method 2540 D) (APHA et al. 1995).

TP concentration was determined using the stannous chloride ammonium molybdate colorimetric method after a persulfate digestion (Standard Method 4500 P B; D). To determine SRP concentrations, runoff and rainfall samples were filtered with a 0.45-µm filter into a plastic vial and stored in a refrigerator (Standard Method 4500 P A). TP and SRP concentrations were measured using a Technicon AutoAnalyzer (APHA et al. 1995). The method detection limit was 1 µg/L.

Results and Discussion

Stormwater Retention and Storage Capacity

During the study period, total monthly rainfall measured on the vegetated roof was 28.2, 136.4, 72.2, 113.2, and 113.0 mm for June, July, August, September, and October, respectively (Table 1). Total event precipitation ranged from 0.8 to 20.8 mm for the 18 rain events monitored from June 2, 2006 to October 22, 2006. Retention of stormwater on the vegetated roof varied from 0.0 to 100.0%, and from 0 to 23.0% on the control roof (Table 1). Overall, the vegetated roof retained 41.5% (64.5 mm

TABLE 1. Stormwater retention in the Waterloo vegetated and control roofs

Storm event (date in 2006)	*Storm Size (mm)*	*Runoff*		*Retention*				*Difference: stormwater retention (%)*
		Vegetated roof (mm)	*Control roof (mm)*	*Vegetated roof (mm)*	*Control roof (mm)*	*Vegetated roof (%)*	*Control Roof (%)*	
June 2–3	12.6	7.5	12.5	5.1	0.1	40.5	0.8	39.7
June 8	2.6	0	2.3	2.6	0.3	100.0	11.5	88.5
June 19	0.8	0	0.7	0.8	0.1	100.0	12.5	87.5
June 19	0.6	0	0.5	0.6	0.1	100.0	16.7	83.3
June 27	0.8	0	0.8	0.8	0.0	100.0	0.0	100.0
June 28–29	11.0	2.6	10.9	8.4	0.1	76.4	0.9	75.5
July 26	7.8	4.1	7.6	3.7	0.2	47.4	2.6	44.9
August 14	6.8	1.1	5.9	5.7	0.9	83.8	13.2	70.6
August 19	4.0	0.9	3.8	3.1	0.2	77.5	5.0	72.5
August 25	7.6	1.5	6.5	6.1	1.1	80.3	14.5	65.8
September 2–3	13.0	7.9	12.3	5.1	0.7	39.2	5.4	33.8
September 18	17.6	7.3	17.6	10.3	0.0	58.5	0.0	58.5
September 22–23	17.4	16.9	16.6	0.5	0.8	2.8	4.5	-1.7
September 27	20.8	1.5	19.4	17.4	1.4	83.7	6.7	76.9
September 30–October 1	13.8	9.8	13.7	4.0	0.1	29.0	0.7	28.3
October 3	3.4	3.3	3.5	0.1	-0.1	2.9	-2.9	5.9
October 19	5.8	11.3	5.8	-5.5	0.0	-94.8	0.0	-94.8
October 22	9.2	16	9.3	-6.8	-0.1	-73.9	-1.1	-72.8
AVERAGE	8.6	5.2	8.4	3.5	0.3	47.6	4.7	42.9
TOTALS: June	28.4	10	27.6	18.4	0.8	64.8	2.8	62.0
July	7.8	4.1	7.6	3.7	0.2	47.4	2.6	44.9
August	18.4	3.5	16.2	14.9	2.2	81.0	12.0	69.0
September	68.8	33.1	66.8	35.7	2.0	51.9	2.9	49.0
October	32.2	40.4	32.3	-8.2	-0.1	-25.5	-0.3	-25.2
Sample Period	155.6	91.1	150.5	64.5	5.1	41.5	3.3	38.2

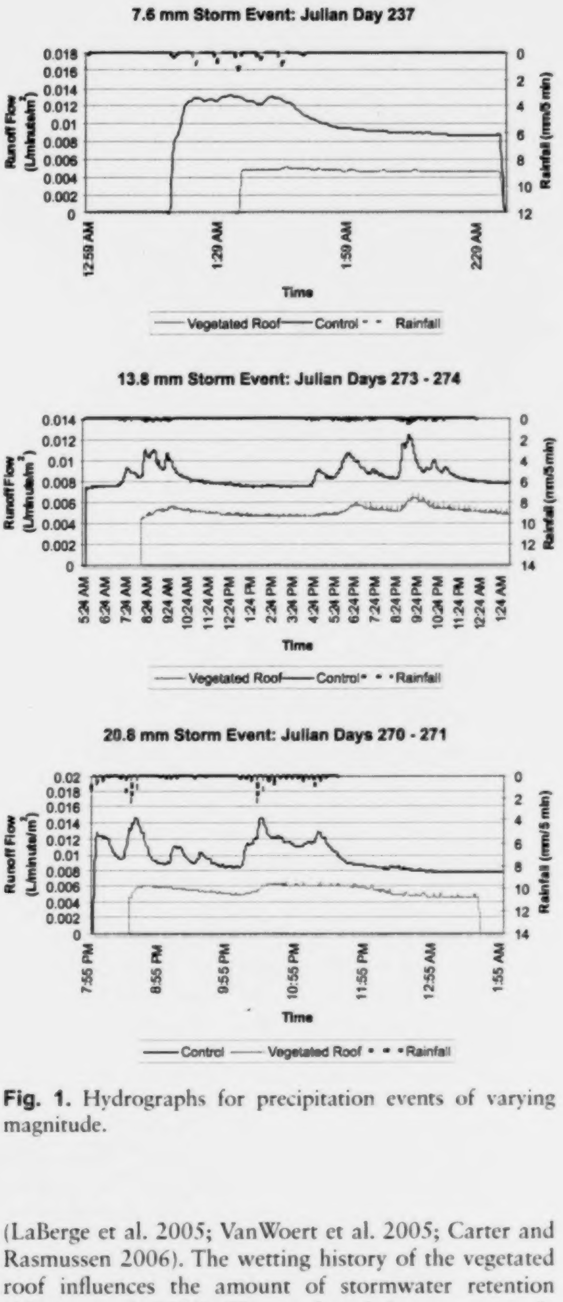
of 155.6 mm) of total rainfall, while the control roof retained 3.3% (5.1 mm of 155.6 mm) of total rainfall; a difference of 38.2% (59.4 mm).

During storm events, storage capacity (the volume of water retained) of the vegetated roof ranged from 0 to 17.4 mm, and from 0 to 1.4 mm for the control roof. The mean vegetated roof storage capacity was 3.5 mm, and mean stormwater retention was 47.6%. The mean control roof storage capacity was 0.3 mm and mean stormwater retention was 4.7%. This represents an increase in average storage capacity and stormwater retention by the vegetated roof of 3.1 mm (42.9%). On three of four occasions, the vegetated roof retained 100% of rainfall during the month of June when a minimum of five antecedent dry days preceded the rain events. The largest storm event to be completely retained was 2.6 mm. Negative retention rates (runoff volume exceeds rainfall input volume) for the vegetated roof (-25.5%) and the control roof (-0.3%) were observed during the month of October when evapotranspiration rates were low and the vegetated roof was saturated from previous rain events.

Stormwater retention varied with storm size. Increases in storm size are related to in increases in vegetated roof storage capacity, while increases in control roof storage capacity were smaller. For storm events ≤3.5 mm, the vegetated roof mean storage capacity was 1.0 mm, and for storm events >3.5 mm, storage capacity increased to 4.4 mm. The control roof storage capacity only increased slightly when rain events exceeded 3.5 mm, with mean storage capacity increasing from 0.1 mm to 0.3 mm.

Several studies have reported mean or total storm water retention rates ≥50% (Jennings et al. 2003; Liu 2003; Connelly and Liu 2005; DeNardo et al. 2005; LaBerge et al. 2005; Liu and Minor 2005; Moran et al. 2005; Carter and Rasmussen 2006; Mentens et al. 2006; TRCA 2006). The higher retention rates reported in the literature compared with the measured retention rates of the Waterloo vegetated roof could be due to several reasons such as: growth medium thickness, substrate composition, rainfall distribution, temperature variability. VanWoert et al. (2005) noted that increasing growth medium thickness increases storm water retention rates. The Waterloo vegetated roof growth medium was 35-mm thick and had a 12-mm water retention fleece (total thickness, 47 mm) while the majority of previous studies were conducted on vegetated roof growth mediums with a thickness ranging from 75 to 150 mm. Results of the Waterloo study are comparable to a 20-mm vegetated roof in Michigan (Monterusso et al. 2004) and a 30-mm vegetated roof in Sweden (Bengtsson et al. 2005).

In addition to the physical and biological characteristics, several factors, including storm size, wetting history, and seasonality, influence stormwater retention of a vegetated roof. The Waterloo vegetated roof showed an inverse relationship between storm size and stormwater retention ($p < 0.05$), which is consistent with the observations of several previous studies


7.6 mm Storm Event: Julian Day 237

13.8 mm Storm Event: Julian Days 273 - 274

20.8 mm Storm Event: Julian Days 270 - 271

Fig. 1. Hydrographs for precipitation events of varying magnitude.

(LaBerge et al. 2005; VanWoert et al. 2005; Carter and Rasmussen 2006). The wetting history of the vegetated roof influences the amount of stormwater retention (Moran et al. 2005). The results of the present study show increased stormwater retention with an increasing number of antecedent dry days ($p < 0.05$) and are consistent with observations of Jennings et al. (2003) and Carter and Rasmussen (2006). Seasonal variability in rainfall, temperature, and evapotranspiration influenced the amount of stormwater retained on the Waterloo vegetated roof. Frequent rainfall, low temperatures, and lower rates of evapotranspiration during fall and winter months reduced retention rates as previously reported by Liu (2003).

Lag Time, Peak Flow, and Runoff Flow Time

Hydrographs for precipitation events of varying magnitudes are presented to compare the lag time, peak flow, and runoff flow time for the Waterloo vegetated and control roofs (Fig. 1). The hydrographs demonstrate that the vegetated roof markedly increased runoff lag time, decreased runoff peak flow, and increased runoff release time relative to the control roof. The mean lag time for the vegetated roof was 74 minutes compared with 15 minutes for the control roof, which represents an increase in lag time by the vegetated roof of 79.7%. The mean peak flow of the vegetated roof was 0.0056 L/minute/m^2, and the mean control roof peak flow was 0.0124 L/minute/m^2. This translates to a reduction in peak flow of 54.8% or 0.0068 L/minute/m^2 by the vegetated roof. With decreased flow rates, the vegetated roof would sometimes increase the runoff release time by several hours. Average vegetated roof flow time was 26.02 h, and 19.8 h for the control roof, which constitutes an increase in average flow time by the vegetated roof of 6.2 h.

Overall, the rainfall response of the vegetated roof did not fluctuate as much as the control roof during a range of rainfall conditions. An increase in storm size and rain intensity showed greater changes in rainfall response from the control roof characterized by a reduction in lag time and an increased runoff peak flow. The rainfall response from the vegetated roof varied to a lesser degree because of the reduction in lag time and increase in peak flow.

Rainfall responses varied with storm size and roof type. Increases in storm size did not impact runoff lag time. However, storm size had an impact on peak flow from the control roof (the flow rate ranged from 0.0081 to 0.0140 L/minute/m^2) but not on the vegetated roof (the flow rate ranged from 0.0047 to 0.0063 L/minute/m^2). During rain events ≤3.5 mm, mean peak flow of the control roof was 0.0103 L/minute/m^2, while it was 0.0048 L/minute/m^2 for the vegetated roof. During rain events >3.5 mm, mean peak flow from the control roof was 0.0127 L/minute/m^2, and 0.0058 L/minute/m^2 from the vegetated roof. This represents a mean peak flow reduction of 54.3% by the vegetated roof. Storm size also influenced runoff flow time. Larger storms increased runoff flow times from both roofs. Storm events ≤3.5 mm had an average runoff flow time from the control roof of 7.2 h, and 7.97 h from the vegetated roof. For storm events >3.5 mm, average runoff flow time from the control roof was 21.9 h, and 29.03 hrs from the vegetated roof.

Several factors including soil moisture, substrate depth, storm size, rain intensity, air temperature, and relative humidity influence lag times and the magnitude of peak flow in vegetated roofs (TRCA 2006). Compared with the control roof, the runoff lag time of the Waterloo vegetated roof significantly increased ($p < 0.05$), and the results were comparable to several other studies (Jennings et al. 2003; DeNardo et al. 2005; Liu and Minor 2005; Moran et al. 2005; VanWoert et al. 2005). Lag time decreased significantly ($p < 0.05$) with increasing rainfall intensity for the Waterloo vegetated and control roof. Compared with the control roof, the peak runoff flow was significantly reduced ($p < 0.05$) in the Waterloo vegetated roof. The increase in vegetated runoff flow time was due to the dampening effect on rainfall infiltration through the multiple layers of the vegetated roof system.

Vegetated Roof Water Quality Treatment

Despite the potential of vegetated roofs to improve stormwater quality (Johnston and Newton 1996; Peck et al. 1999), relatively few studies have examined the water quality treatment performance of vegetated roofs in Canada. The first studies of green roof water quality were conducted in Germany (VanWoert et al. 2005; Getter and Rowe 2006). Subsequent studies report vegetated roofs as a source of phosphorus and show they may not be an effective technology for metals removal in stormwater (Berndtsson et al. 2006; TRCA 2006). In the following, treatment performance of the Waterloo vegetated roof for selected water quality parameters are presented and compared with the results of previous studies.

Total Dissolved Solids

The average total dissolved solids (TDS) in vegetated roof runoff was significantly greater than in either rainfall or control roof runoff ($p < 0.05$). No other previous studies have reported TDS concentrations in vegetated roof runoff. The elevated TDS concentrations are likely related to the presence of dissolved materials from both the vegetated roof growth medium and fertilizer application.

The highest concentration of TDS in runoff was measured from the vegetated roof during the study period. The mean TDS concentration in runoff from the vegetated roof was 0.131 mg/L with a range of 0.036 to 0.235 mg/L. The mean TDS concentration in control roof runoff was 0.035 mg/L with a range of 0.003 to 0.144 mg/L. The mean TDS concentration of rainfall was 0.013 mg/L and ranged from 0.009 to 0.024 mg/L. During individual storm events, TDS concentrations in vegetated roof runoff typically increased, and control roof runoff concentrations decreased over the sampling period of the storm event.

Suspended Solids

Temporal variability of SS measured in rainfall and runoff from the vegetated and control roof is shown in Fig. 2. Average SS concentration in the control roof was 8.3 mg/L with a range of ≤0.1 to 66.0 mg/L over the sample period. The vegetated roof had a lower mean SS concentration of 5.6 mg/L with a range of ≤0.1 to 15.0 mg/L. Average vegetated roof SS concentration was 32.5% (2.7 mg/L) less than the control roof average SS concentration. Rainfall had the lowest mean SS concentration of 2.3 mg/L, which ranged from ≤0.1 to 6.5 mg/L.

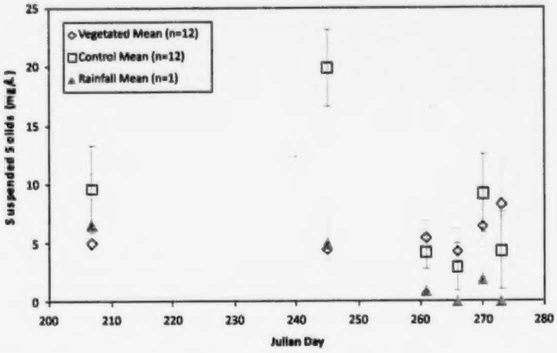


Fig. 2. Suspended solids concentration (mean ± standard deviation) in rainfall and runoff from the vegetated and control roofs.

Total suspended solids (TSS) concentrations were lower in runoff from the vegetated roof than the control roof, but not significantly different. Levels of TSS measured in runoff from the Waterloo vegetated roof are comparable to a previous study which reported an 85.4% reduction in TSS concentration in vegetated roof runoff (TRCA 2006). The lower TSS concentrations in the vegetated roof runoff are likely due to the filter cloth layer in the Waterloo vegetated roof system which is designed to prevent the loss of organic material from the growth medium.

Total and Soluble Reactive Phosphorus

Temporal variation in TP concentration measured in the rainfall and in runoff from a vegetated and a control roof is illustrated in Fig. 3. The mean TP concentration of vegetated roof runoff was over five times greater than that in rainfall, and four times greater than in runoff from the control roof. The average TP concentration of the vegetated roof runoff was 99.8 µg/L, with a range from 33.8 to 204.8 µg/L. The average TP concentration in runoff from the control roof was 15.4 µg/L and ranged from 1.0 to 102.9 µg/L. In rainfall, the average TP concentration was 16.9 µg/L and ranged from 4.5 to 33.3 µg/L. During individual storm events, TP concentrations from the vegetated and control roofs varied temporally over the storm event. Concentrations of TP in runoff from the vegetated roof were quite variable depending upon the timing and duration of the storm event. During storm events, TP concentrations in runoff from the control roof fluctuated but typically decreased over time.

Temporal variation in SRP measured in the rainfall and in runoff from a vegetated and a control roof is illustrated in Fig. 4. In runoff from the vegetated roof, the mean SRP concentration was 40.0 µg/L, with a range from 7.7 to 98.0 µg/L. A majority of runoff samples from the control roof had SRP concentrations below the method detection limit of 1 µg/L. The mean SRP concentration of the control roof runoff was 3.8 µg/L, with a range from 1 to 12.5 µg/L. The concentration of SRP in rainfall was below the detection limit of 1 µg/L with the exception of one storm event when it was 2 µg/L. Overall, SRP concentrations in runoff from the vegetated roof were ten times greater than that measured in runoff from the control roof, and twenty times greater than concentrations measured in rainfall.

TP and SRP concentrations in runoff from the Waterloo vegetated roof were significantly greater than in either the rainfall or the control roof runoff ($p < 0.05$). Levels of TP measured in the present study exceed the provincial water quality objective for TP (30 µg/L; OMEE 1999). However, concentrations of TP in the control roof were comparable to levels measured in rainfall and were well below the limits set by the provincial water quality objectives. The results of the present study are comparable to previous studies (Moran et al. 2005; Berndtsson et al. 2006; TRCA 2006) and demonstrate that the Waterloo vegetated roof is a source of both TP and SRP.

The primary source of phosphorus in the vegetated roof is likely related to the biogeochemical nature of the growth medium and the application of fertilizer (Jennings et al. 2003; Berndtsson et al. 2006; TRCA 2006). The Waterloo vegetated roof growth medium consisted of 14% organic material, and fertilizers (3.5% P_2O_2) were applied June 9, 2006 (Xero Flor 2006b). Fertilizers were used in the maintenance of the Waterloo vegetated roof to help establish plant growth and coverage (Berndtsson et al. 2006). Both TRCA (2006) and Berndtsson et al. (2006) recommend the use of slow release fertilizers to limit nutrient input and to reduce phosphorus leaching. Older established vegetated roofs with limited nutrient input have been shown to retain phosphorus. In Germany, Köhler and Schmidt (2003) documented phosphorus retention of 67% by a 15-year-old vegetated roof. In addition, studies indicating the potential of a vegetated roof as a phosphorus source have reported decreases in phosphorus concentrations over time. TRCA (2006) reported a 214% decrease in phosphorus from a vegetated roof over a one year study period.

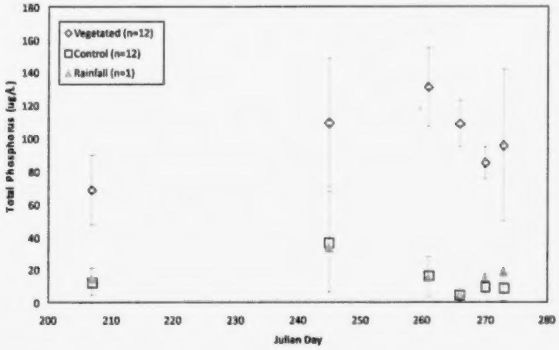


Fig. 3. Total phosphorus concentration (mean ± standard deviation) in rainfall and runoff from the vegetated and control roofs.

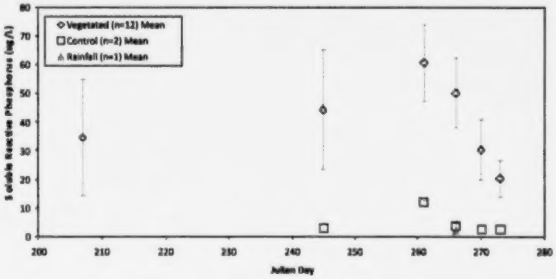


Fig. 4. SRP concentration (mean ± standard deviation) in rainfall and runoff from the vegetated and control roofs.

Conclusions

The wet weather performance of an extensive vegetated roof was evaluated in Waterloo, Ontario. The results of the study provide a better understanding of the utility of vegetated roofs as a source control best management practice for stormwater. The Waterloo vegetated roof had an average stormwater retention rate of 47.6% compared with 4.7% in the control roof. Mean storage capacity of the vegetated roof was over ten times greater than for the control roof.

Storm size, the number of antecedent dry days between storms, and seasonal changes in temperature, rainfall, and evapotranspiration influenced stormwater retention rates. The vegetated roof had an increased lag time, reduced peak flow, and extended runoff flow time compared with the control roof. Rain intensity and storm size influenced rain responses from both roof types. The vegetated roof did not significantly improve water quality and was a source of TP and SRP compared with the control roof. Phosphorus loss from the vegetated roof exceeded limits set by provincial water quality objectives. Sources of TP and SRP in vegetated roof runoff are likely due to the organic content present in the vegetated roof growth medium and fertilizer application. The Waterloo study demonstrates the utility of vegetated roofs as a source control measure to enhance stormwater retention. The application of vegetated roofs should be considered particularly when new buildings are being designed for infill projects in urban areas with little stormwater infrastructure.

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Evaluation of Green Roofs for Runoff Retention, Runoff Quality, and Leachability

Tim Van Seters,* Lisa Rocha, Derek Smith, and Glenn MacMillan

Toronto and Region Conservation Authority, 5 Shoreham Drive, Downsview, Ontario M3N 1S4

This three-year study evaluates the quantity and quality of runoff from an extensive green roof on a multistory building in Toronto. Laboratory physical, chemical, and leachate analyses of eleven commercially available green roof growing media were also undertaken to help identify the potential influence that the growing media may have on runoff chemistry. Continuous precipitation and runoff data collected over 18 months outside of the winter period indicated that the green roof discharged 63% less runoff than a neighbouring conventional modified bitumen roof. Runoff volumes from the green roof averaged 42% less than the conventional roof in April and November, and between 70 and 93% less during the summer months. Water samples were collected from both roofs during 21 rain events in 2003 and 2004 and analyzed for general chemistry (e.g., pH, total suspended solids), metals, nutrients, bacteria (n = 16), and polycyclic aromatic hydrocarbons (n = 18). Loads of most chemical variables in green roof runoff were lower than from the conventional roof. Exceptions included constituents such as calcium, magnesium, and total phosphorus, which were either naturally present in the media or were added to promote plant growth. Total phosphorus concentrations in green roof runoff were significantly higher than the conventional roof (α = 0.001), and regularly exceeded the Ontario receiving water objective (0.03 mg/L). Phosphorus concentrations fell significantly after the first year of monitoring (α = 0.001), suggesting that the nutrient is being leached from the media. Chemical analyses of green roof growing media showed that levels of most constituents were similar to or lower than typical background concentrations for agricultural soils in Ontario. However, leachate concentrations from several media exceeded receiving water standards for phosphorus, aluminum, copper, iron, and vanadium. This study highlights the importance of engineering green roof media to minimize leaching of nutrients and other contaminants while maintaining their ability to support plant growth.

***Key words*: green roof, phosphorus, growing media, runoff quality**

Introduction

Controlling runoff at source has become a cornerstone of good stormwater management practice both in older built-up areas and new developments within the Greater Toronto Area (GTA). Green roofs offer significant advantages over other source controls because they can be installed in dense urban areas where space for structural practices is not available, and they function well in areas where low permeability soils may limit the effectiveness of stormwater infiltration technologies such as bioretention areas or permeable pavements. In addition to their ability to retain stormwater, and their obvious aesthetic qualities, green roofs offer numerous advantages over traditional roofs, including energy conservation (Onmura et al. 2001; Liu and Baskaran 2005), mitigation of the urban heat island effect (Bass et al. 2003; Wong et al. 2003), and improved urban biodiversity (Brenneisen 2003). However, these advantages come at a cost as green roofs in North America are typically more expensive to construct than conventional roofs, and structural modifications may be needed to support the additional weight of a vegetated roof (TRCA 2007). To help overcome this barrier, some GTA municipalities have implemented or are currently exploring the possibility of implementing green roof incentive programs.

Most research on the stormwater management benefits of green roofs has focused on the ability of green roofs to attenuate peak flows and reduce the total volume of stormwater runoff by retaining rain water in the growing media. In a comprehensive review of international green roof literature on rainfall runoff relationships, Mentens et al. (2006) reported a median annual retention rate of 45% for extensive green roofs with substrate depths between 3 and 14 cm (median: 10 cm). Most North American studies indicate runoff retention during the growing season of at least 60% on roofs with gentle slopes (<10%) and growing medium depths of 6 cm or more (Hutchinson et al. 2003; Rowe et al. 2003; Carter and Rasmussen 2006; Getter et al. 2007; Hathaway et al. 2008). These rates typically decline to roughly 40% on growing media between 2- and 4-cm deep (Liesecke 1998; Russell and Schickedantz 2003), with some exceptions (Rowe et al. 2003). Getter et al. (2007) reported a 10% rise in mean retention associated with a decrease in roof slope from 25 to 2%. Retention of rainfall has also been shown to extend the duration of flow and reduce peak flow rates by between 50 and 85% (e.g., Hutchinson et al. 2003; Johnston et al. 2004; Liu and Minor 2005; Hathaway et al. 2008), thereby reducing the erosive power of stream flows.

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* Corresponding author: tvanseters@trca.on.ca

In studies of the stormwater management benefits of green roofs, the quality of runoff has received much less attention than the volume and rate of runoff. The main sources of contaminants on roofs are atmospheric deposition and leaching from roofing materials. Several researchers have shown that conventional roof runoff may contain elevated levels of polycyclic aromatic hydrocarbons (PAHs), organic halogens, and heavy metals such as lead, copper, zinc, and cadmium (Thomas and Greene 1993; Clark et al. 2001; Van Metre and Mahler 2003). Green roofs would normally be expected to discharge fewer of these contaminants than conventional roofs, but there are few published studies that have quantified this difference for metals, hydrocarbons, and other organic compounds.

In a field study of a green roof in Germany, Forster and Knoche (1999) reported higher heavy metal concentrations in green roof runoff than in rainwater, but lower concentrations of PAHs. The difference in metal concentrations was attributed to leaching from unprotected metal surfaces around the roof. Studies of nutrients in green roof runoff in North Carolina (Hathaway et al. 2008) and Oregon (Hutchinson et al. 2003) identified phosphorus as a potential concern. In both studies, green roof runoff concentrations of phosphorus were well above receiving water standards. However, Hathaway et al. (2008) did not find a statistically significant difference between green roof and conventional roof nutrient loads. Composted cow manure, which made up 15% of the growing medium, was identified as a likely source of nutrients in the North Carolina study. In a study of four extensive green roofs in southern Sweden, Berndtsson et al. (2006) also found elevated levels of phosphorus, which they suggested was likely a result of fertilizer application. The green roofs in this study were also a source of zinc, copper, and lead, although concentrations were well below those typical of urban runoff.

In a follow-up to the North Carolina study, Hunt et al. (2006) postulated a relationship between nutrient concentrations in runoff and the type and quantity of compost in the growing media. Concentrations were particularly elevated when the compost contained cow manure. In the same study, laboratory testing of 3 growing media showed a decrease in total nitrogen with 10 applications of synthetic rainwater over a 10-week period, but there was no trend in total phosphorus. Sampling at field sites also showed no statistically significant trend in phosphorus or nitrogen compounds.

The primary purpose of the present study was to evaluate the impact of rooftop green roofs on the quantity and quality of stormwater runoff from a typical green roof installation under climate conditions characteristic of the GTA. Laboratory physical, chemical, and leachate analyses of several commercially available green roof growing media were also undertaken to help identify potential concerns about the quality of green roof runoff.

Materials and Methods

Study Site

The monitoring study was conducted on the roof of a multistory building at York University in the City of Toronto (Fig. 1). The study area consisted of two surfaces: (i) a 131-m^2 shingled, modified bitumen roof, hereafter referred to as the conventional roof, and (ii) a 241-m^2 green roof vegetated with wildflowers. Both roof surfaces had a 10% slope. Installed in 2002, the 14-cm growing medium on the green roof was composed of crushed volcanic rock, compost, blonde peat, cooked clay, and washed sand. It was designed to be light weight, retain water, and resist compaction. The green roof irrigation system came on every night during the first summer (June to October), and thereafter, when soil moisture content fell below 20%, or on average, once every two days.

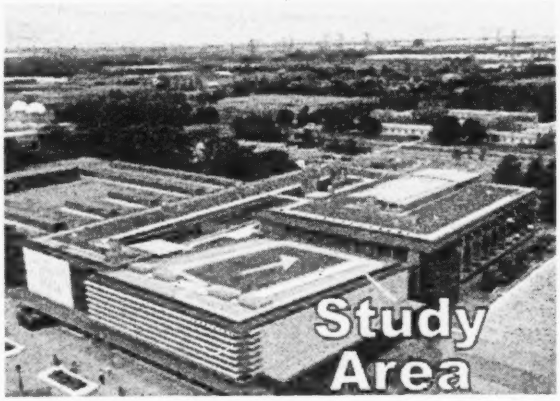

Fig. 1. Study area on the York University Computer Science Building.

Field Monitoring

Figure 2 shows the location of monitoring instruments. Measurements included precipitation, flow, water quality, soil moisture, relative humidity, air temperature, and the temperature of the growing medium. Precipitation at the site was measured using a tipping bucket rain gauge. Nearby Atmospheric Environment Service (AES) stations provided back-up data on rain or snow conditions when site data were not available (Environment Canada 2006).

The green roof and conventional roof drained to separate eaves troughs at the bottom of the sloped study area. Runoff from the green roof and conventional roof were monitored continuously with two magnetic induction instruments (MAGmeters). Flow rates were recorded at 1 minute intervals to a measurement accuracy of ±0.5%.

Green roof and conventional roof runoff samples were collected for water quality analysis using two automated water samplers. The sampling interval was selected based on the average duration of runoff. The conventional roof and green roof samplers collected samples at 5 and 10 minute intervals, respectively. Atmospheric deposition during wet and dry weather was monitored using an open polyethylene bag lining a 48-cm diameter bucket. Samples were collected after rain events. All samples were preserved and submitted according to Ontario Ministry of the Environment (OMOE) guidelines (OMOE 2003) and analyzed at OMOE laboratories in Toronto. The runoff samples were analyzed for general chemistry (e.g., total suspended solids [TSS], alkalinity), nutrients (phosphorus and nitrogen compounds), metals, bacteria, and PAHs. The precipitation samples were analyzed for general chemistry and nutrients and metals.

Samples were analyzed in the laboratory following principles outlined in *Standard Methods* (APHA et al. 1998). Briefly, metals were unfiltered totals analyzed by inductively coupled plasma atomic emission spectroscopy (ICP-AES). Nutrients were determined by colourimetry. *Escherichia coli* was determined through membrane filtration. TSS was determined by gravimetry using a 1.5- to 2.0-micron glass fibre filter. Analysis of PAHs was conducted by liquid/liquid extraction followed by gas chromatography–mass spectrometry.

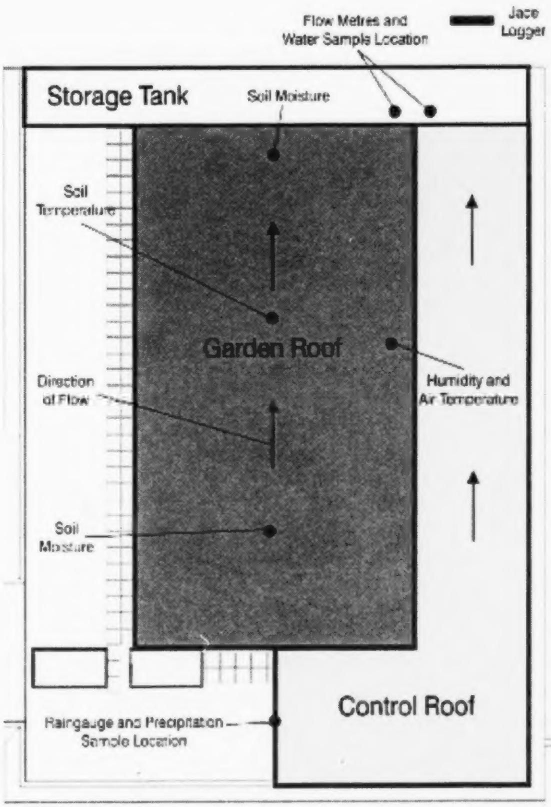

Fig. 2. Schematic of monitoring setup.

Data Analysis

The water quality data were graphically analyzed and found to be log normally distributed; therefore, the mean and confidence intervals were performed on log-transformed data sets. Concentrations of variables below laboratory detection limits were represented by half the detection limit. Confidence intervals are not presented for variables with detection frequencies of less than 50% due to the bias caused by the substituted data. Detection frequencies provide the basis for comparison of these variables.

Unit area loads from the conventional roof and green roof were calculated separately for the sampled and unsampled events, and subsequently summed to arrive at an overall load calculation for the two years of water quality sampling (2003 to 2004). The sampled event loads represented the sum of the individual loads for each sampled event ($n = 21$). For sampled events, each individual event load represents the event runoff volume multiplied by the sampled concentration for the same event. The unsampled event loads for each of the two sampling years (2003 and 2004) were calculated as the total measured runoff volume for the unsampled events multiplied by the mean concentration for the sampled events. This method assumes that the mean concentration for the sampled and unsampled events is similar, which is reasonable given that the variables analyzed did not exhibit a relationship between event concentration and event size. Including the unsampled events allowed for several small rain events, for which there was little or no green roof runoff, to be represented in the overall load calculation.

Analyses of Green Roof Growing Media

Chemical (bulk soil and leachate) and physical analyses were conducted on 11 green roof growing media currently available commercially in order to determine the potential impact of growing media constituents on runoff chemistry. Analyses of the physical and chemical quality of growing media were conducted by a certified private laboratory (Entech) in Mississauga, Ontario. Variable groups analyzed included general chemistry, metals (EPA 3050B/200.7), and nutrients (EPA 351.2/365.1) using standard United States Environmental Protection Agency methods (U.S. EPA 1983). Grain size distribution was determined by passing the samples through a 2-mm screen. The material that did not pass through the screen was deemed to be gravel. The material that passed through the screen was analyzed using a hydrometer to determine the percentages of sand, silt, and clay that made up the remaining portion of the sample.

Leachate tests were also conducted on the 11 growing media. These tests were not intended to mimic actual runoff quality from an established green roof since the media samples were not planted or layered in the same way as they would have been on an actual roof.

Instead, the purpose of the tests was to identify the types of pollutants that may be leached from the media, their concentration relative to surface water criteria, and the extent to which leachability of pollutants may vary among the growing media. Each media sample was leached four times in order to identify the chemical constituents that may be susceptible to leaching, and the general direction in which pollutant concentrations may be expected to change over time.

The laboratory leaching procedure was based a standard waste management leachate method (EPA 1311) (U.S. EPA 1996) with minor modifications to suit the objectives of this study. A minimum of 100 grams of each sample was placed in an extractor vessel made of inert material (polytetrafluoroethylene for inorganics, type 316 stainless steel for organics), and a quantity of reagent water (pH of 6) equal to 20 times the weight of the solid phase was added. The vessel was placed in a rotary agitation device and rotated end-over-end at 30 ± 2 rpm for 18 ± 2 hours, with ambient temperature maintained at 23 ± 2°C. For media with particles greater than 1 cm in its narrowest dimension, particle size reduction was performed prior to agitation. Following agitation, the liquid and solid phases were separated using a borosilicate glass fibre filter with a pore size of 0.6 to 0.8 µm. This process was repeated on the same sample three more times. The first and fourth liquid extracts were collected and analyzed for general chemistry, nutrients (EPA 353.1, 353.2, 350.1; EPA 365.1) (U.S. EPA 1983), and metals (EPA 200.7, 200.15, 3005) (U.S. EPA 1983, 1994, 1996 [respectively]) by Entech laboratories.

Results

Hydrology

Runoff and precipitation were monitored during 154 runoff events from May to November 2003, June to November 2004, and April to August 2005. An event consisted of all rainfall from the start of the storm until runoff from the conventional and green roofs ceased, including those for which there was no green roof runoff. Data were not available during the winter from January to March.

Compared with regional climate normals (1971 to 2000), rainfall over the monitoring period was 19% greater and temperatures were on average 2.3°C warmer. The 2003 and 2005 monitoring seasons were the wettest, with roughly 30% more rainfall than the 30 year average. By contrast, the 2004 season was 2% drier and average temperatures were only 0.6°C warmer than the historical average.

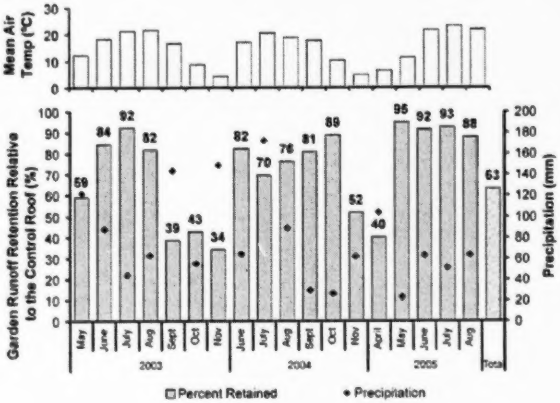

Fig. 3. Mean monthly air temperature, precipitation and the percent of runoff retained by the green roof relative to the conventional roof.

Figure 3 presents air temperature, precipitation, and retention rates by month for the study period. Overall, the green roof discharged 63% less runoff than the conventional roof. Runoff retention rates were lower in 2003 than in later years because of relatively large amounts of precipitation in September and November of that year, and more frequent irrigation, both of which adversely influenced the green roof's capacity to retain water. During 2003, the green roof retained only 54% of precipitation, while in 2004 and 2005 approximately 75% of precipitation was retained. The conventional roof retained 2% of total rainfall over the study period.

Rainfall volumes, evapotranspiration rates, and antecedent moisture content were the key factors explaining monthly and event-by-event variations in green roof retention rates. As the days became shorter and air temperatures dropped, the green roof retained less water. Retention rates for summer events were between 78 and 85%, compared with spring and fall retention rates of between 39 and 64% (Fig. 3). Figure 4 illustrates the declining capacity of the green roof to retain water and reduce peak flows as event size increased. Rainfall in November and April tended to be less effectively retained because evapotranspiration rates are much lower during these months. Retention rates were also generally lower during the second of two back-to-back events due to higher antecedent moisture conditions.

Water Quality

Precipitation and runoff water quality analysis was completed for 21 events from September to November, 2003, and June to November, 2004. Table 1 presents the quality of dry and wet precipitation over the monitoring period. The rain was relatively acidic (pH of 5.9) and contained low levels of dissolved solids. Phosphorus, nitrogen, and metal concentrations were generally low, although maximum values indicate that atmospheric contributions can be a significant source of these constituents.

Table 2 presents detection frequencies, mean concentrations, 95% confidence intervals, and differences between the conventional and green roofs at the indicated significance level. The precipitation was

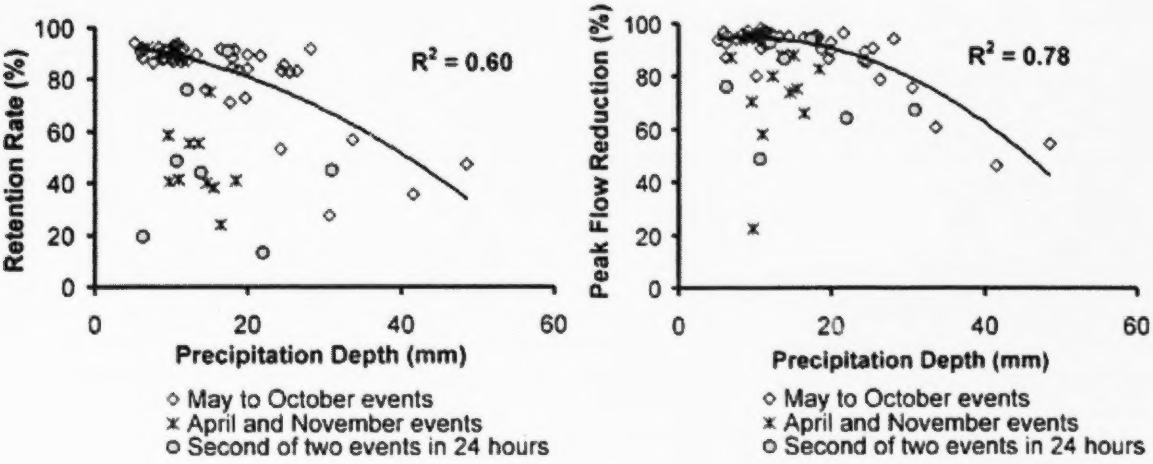

Fig. 4. Relationships between precipitation depths and retention rates for events greater than 5 mm. Trend lines and R^2 values relate to events occurring between May and October.

TABLE 1. Characterization of wet and dry precipitation

Variable	*Units*	*MDL*[a]	*n*	*Minimum*	*Maximum*	*Median*
Conductivity	µS/cm	1	19	7	27	16
pH	none	—	19	4.41	6.86	5.89
Nitrogen; ammonia+ammonium	mg/L	0.002	19	0.10	1.27	0.37
Nitrogen; nitrate+nitrite	mg/L	0.005	20	0.01	1.43	0.34
Nitrogen; TKN	mg/L	0.02	17	0.26	1.52	0.56
Phosphorus; phosphate	mg/L	0.0005	18	0.0003	0.009	0.002
Phosphorus; total	mg/L	0.002	18	0.01	0.04	0.01
Copper	µg/L	1.6	18	4.0	23.2	11.0
Zinc	µg/L	0.6	18	5.4	25.6	11.3
Lead	µg/L	5	18	<MDL	12.20	<MDL

[a] MDL = reporting method detection limit.

chemically altered as it filtered through the green roof growing media, resulting in higher concentrations of major ions (e.g. Mg, Ca, S, Cl, Na), pH, and hardness (as $CaCO_3$) relative to the conventional roof. Runoff from both roofs contained very low levels of turbidity and TSS. Detection frequencies of lead, nickel, cadmium, molybdenum, and beryllium were greater in green roof samples, but concentrations were below Ontario receiving water guidelines (OMOE 1994). Runoff from the conventional roof had higher mean concentrations of copper ($\alpha = 0.001$), zinc ($\alpha = 0.1$), and manganese ($\alpha = 0.001$), and higher PAH detection frequencies. While zinc levels were generally low, concentrations of copper in runoff from both roofs regularly exceeded the Ontario receiving water guideline.

Green roof growing media are typically engineered to include nutrients in order to promote plant growth. This is evident from the much higher levels of phosphorus, total Kjeldahl nitrogen (TKN), and potassium in green roof runoff. Levels of other nitrogen species (nitrate, nitrite, ammonia) were higher in runoff from the conventional roof. Total phosphorus is of particular concern since green roof concentrations were several times higher than the Ontario receiving water standard of 0.03 mg/L, established to protect against excess algae and plant growth in rivers and streams.

Unit area loads from the two roofs are presented in Table 3. Conventional roof loads of most chemical constituents of concern were considerably greater than green roof loads, which was to be expected since the green roof generated much less runoff. Even variables with higher concentrations in green roof runoff, such as TKN, vanadium, and nickel, were discharged in larger quantities (by mass) from the conventional roof. By contrast, green roof loads of total phosphorus were more than triple that of the conventional roof over the monitoring period. Most of the phosphorus was transported as phosphate, which is more biologically

TABLE 2. Mean concentrations and 95% confidence limits for green roof (G) and conventional roof (C) runoff samples

Variable (units)	*MDL*[a]	*PWQO*[b]	*n*	*Green roof*				*Conventional roof*				*Comparison, significance level*[d,e]
				% >MDL	*Geo mean*	*LL*[c]	*UL*[c]	*% >MDL*	*Geo mean*	*LL*	*UL*	
Chloride (mg/L)	0.2		21	100	9.0	6.8	11.8	95	1.0	0.7	1.4	G>C (0.001)
Mercury (µg/L)	0.02	0.2	19	0	<MDL	–	–	5	<MDL	–	–	N/A
Sodium (mg/L)	0.1		21	100	6.1	5.6	6.7	100	1.1	0.9	1.2	G>C (0.001)
Potassium (mg/L)	0.05		21	100	6.6	5.6	7.8	100	1.9	1.7	2.1	G>C (0.001)
Hardness as CaCO$_3$ (mg/L)	1		21	100	86.5	77.2	97.0	100	15.3	13.3	17.5	G>C (0.001)
Sulphate (mg/L)	2.5		21	100	19.4	15.6	23.9	90	4.3	3.4	5.5	G>C (0.001
TSS (mg/L)	2.5		21	33	<MDL	–	–	95	5.8	4.4	7.6	N/A
Conductivity (µS/cm)	1		18	100	209	187.9	231.9	100	45	39.6	50.6	G>C (0.001)
pH (none)	none	6.5–8.5	18	100	8.10	7.9	8.3	100	7.26	7.2	7.3	G>C (0.001)
Alkalinity (mg CaCO$_3$/L)	2.5		18	100	62.0	56.7	67.7	100	13.7	12.3	15.3	G>C (0.001)
Turbidity (FTU)	0.01		10	100	1.22	0.90	1.66	100	2.61	1.96	3.48	C>G (0.05)
Nitrogen; nitrate (mg/L)	0.005		21	100	0.156	0.11	0.23	100	0.412	0.31	0.56	C>G (0.01)
Nitrogen; nitrite (mg/L)	0.001		21	100	0.019	0.02	0.02	100	0.029	0.02	0.03	C>G (0.05)
Nitrogen; NH$_3$ + NH$_4$ (mg/L)	0.002		21	80	0.008	0.00	0.02	100	0.213	0.16	0.29	C>G (0.001)
Nitrogen; TKN (mg/L)	0.02		21	100	1.110	0.88	1.41	100	0.616	0.49	0.77	G>C (0.05)
Total phosphorus (mg/L)	0.0005		21	100	0.318	0.23	0.45	90	0.053	0.04	0.07	G>C (0.001)
Phosphate (mg/L)	0.002	0.03[f]	21	100	0.241	0.16	0.36	100	0.009	0.005	0.019	G>C (0.001)
Escherichia coli (CFU/100mL)		100	16	100	113	34.4	372.2	100	25	9.5	66.8	no diff. (0.1)
Aluminum (µg/L)	11	75[f]	20	100	47.7	42.7	53.3	100	58.2	46.7	72.6	no diff. (0.1)
Barium (µg/L)	0.2		20	100	13.2	11.7	14.8	100	3.3	2.7	3.9	G>C (0.001)
Beryllium (µg/L)	0.02	11	20	50	<MDL	–	–	5	<MDL	–	–	N/A
Calcium (mg/L)	0.005		20	100	20.53	18.07	23.32	100	4.59	4.01	5.25	G>C (0.001)
Cadmium (µg/L)	0.6	0.1[f]	20	25	<MDL	–	–	13	<MDL	–	–	N/A
Cobalt (µg/L)	1.3	0.9	20	10	<MDL	–	–	13	<MDL	–	–	N/A
Chromium (µg/L)	1.4	8.9	20	5	<MDL	–	–	14	<MDL	–	–	N/A
Copper (µg/L)	1.6	1 or 5[f,g]	20	100	44.8	33.5	59.8	100	108.1	86.4	135.3	C>G (0.001)
Iron (µg/L)	0.8	300	20	100	25.5	19.8	32.9	100	34.2	25.9	45.2	no diff. (0.1)
Magnesium (mg/L)	0.008		20	100	4.16	3.3	5.2	100	0.73	0.6	0.9	G>C (0.001)
Manganese (µg/L)	0.2		20	100	1.41	1.2	1.7	100	8.33	6.6	10.4	C>G (0.001)
Molybdenum (µg/L)	1.6	40	20	15	<MDL	–	–	0	<MDL	–	–	N/A
Nickel (µg/L)	1.3	25	20	60	1.3	1.0	1.6	15	<MDL	–	–	N/A
Lead (µg/L)	10	1–5[f,h]	20	30	<MDL	–	–	0	<MDL	–	–	N/A
Strontium (µg/L)	0.1		20	100	85.3	77.0	94.6	100	15.2	13.3	17.4	G>C (0.001)
Titanium (µg/L)	0.5		20	25	<MDL	–	–	90	1.17	0.8	1.6	N/A
Vanadium (µg/L)	1.5	6	20	80	1.84	1.5	2.3	25	<MDL	–	–	N/A
Zinc (µg/L)	0.6	20[f]	20	100	6.89	5.4	8.8	100	9.85	8.3	11.7	C>G (0.1)
Phenanthrene (ng/L)	10	30	18	39	10.1	–	–	89	60.9	33.7	109.9	N/A
Anthracene (ng/L)	10	0.8	18	0	<MDL	–	–	50	<MDL	–	–	N/A
Fluoranthene (ng/L)	10	0.8	18	28	<MDL	–	–	89	78.7	40.7	152.2	N/A
Pyrene (ng/L)	10		18	28	<MDL	–	–	72	51.7	24.9	107.0	N/A
Benzo(a)anthracene (ng/L)	20	0.4	18	6	<MDL	–	–	56	28.3	18.0	44.3	N/A
Chrysene (ng/L)	10	0.1	18	6	<MDL	–	–	67	31.7	16.9	59.6	N/A
DMBA (ng/L)[i]	10		18	0	<MDL	–	–	0	<MDL	–	–	N/A
Benzo(b)fluoranthene (ng/L)	10		18	6	<MDL	–	–	72	33.3	17.8	62.2	N/A
Benzo(k)fluoranthene (ng/L)	10	0.2	18	11	<MDL	–	–	67	27.1	14.9	49.2	N/A
Benzo(e)pyrene (ng/L)	10		18	6	<MDL	–	–	67	23.7	13.6	41.4	N/A
Benzo(a)pyrene (ng/L)	3		18	22	<MDL	–	–	78	23.4	10.8	50.9	N/A
Perylene (ng/L)	10	0.07	18	0	<MDL	–	–	56	11.4	7.9	16.6	N/A
Indeno(1,2,3-c,d)pyrene (ng/L)	20		18	6	<MDL	–	–	56	32.5	19.4	54.4	N/A
Dibenzo(a,h)anthracene (ng/L)	20	2	18	0	<MDL	–	–	6	<MDL	–	–	N/A
Benzo(g,h,i)perylene (ng/L)	20	0.02	18	0	<MDL	–	–	56	26.8	17.3	41.5	N/A

[a] MDL = reporting method detection limit.

[b] Provincial Water Quality Objective (OMOE 1994).

[c] LL = lower confidence limit; UL = upper confidence limit.

[d] Comparison between green and conventional roofs with significance level reported in parenthesis ($\alpha = x$) for variables with detection frequencies >50%.

[e] n/a = Not applicable.

[f] Interim PWQO.

[g] 1 and 5 µg/L at hardness <20 mg/L and > 20 mg/L, respectively.

[h] 1, 3, and 5 µg/L at hardness <30 mg/L, 30 to 80 mg/L, and >80 mg/L, respectively.

[i] DMBA = 7,12-dimethylbenz(a)anthracene.

available than particulate phosphorus, and thus presents a more serious concern. Over the monitoring period, 86% of green roof total phosphorus loads were in the form of phosphate, compared with only 46% from the conventional roof.

Figure 5 presents time series plots for phosphorus, phosphate, nitrate, and copper. These were the only variables analyzed with mean concentrations in green roof runoff that were significantly different ($\alpha = 0.01$) between the 2003 ($n = 9$) and 2004 ($n = 12$) monitoring seasons. The influence of year-to-year variations in flow volumes, event precipitation depths, and rainfall chemistry on runoff concentrations was investigated as a possible source of observed changes, but regression analysis showed these factors to be insignificant.

The direction of change differed among the variables shown in Fig. 5. While total phosphorus and phosphate concentrations in green roof runoff decreased, nitrate and copper increased. There was no significant difference in mean conventional roof concentrations of total phosphorus, nitrate, and copper, but orthophosphorus concentrations declined ($\alpha = 0.05$). The change in green roof concentrations of phosphorus was much greater than observed for the other variables. Mean 2004 concentrations of total phosphorus (0.18 mg/L) were less than one third of mean concentrations observed in 2003 (0.7 mg/L). Despite lower concentrations in 2004, phosphorus loads from the green roof remained higher than those from the conventional roof.

Growing Media Chemical and Physical Analyses

Eleven samples of green roof growing media were obtained from manufacturers and analyzed for general chemistry, nutrients, and metals. Analyses were conducted to determine the chemical and physical composition of bulk samples, as well as the quality of leachate from each of these media. Table 4 provides manufacturer descriptions of each media tested and the laboratory grain size test results. Composition information is not provided for J

TABLE 3. Unit area loads and percent difference relative to the conventional roof for the 2003 and 2004 monitoring seasons

Variable	*Unit area load (mg/m^2)* [a]		*% Difference (conventional roof versus green roof)*
	Green roof	*Conventional roof*	
Chloride	4,310.2	1,333.7	-223.2
Sodium	2,606.3	1,212.7	-114.9
Potassium	2,463.2	2,091.5	-17.8
Hardness as $CaCO_3$	39,604.2	17,944.5	-120.7
Sulphate	8,890.3	5,310.9	-67.4
Suspended solids	763.2	6,752.0	88.7
Nitrogen; nitrate	40.8	450.1	90.9
Nitrogen; nitrite	7.5	31.4	76.2
Nitrogen; ammonia +ammonium	6.1	245.7	97.5
Nitrogen; total Kjeldahl	575.3	677.7	15.1
Phosphorus; phosphate	207.2	26.1	-694.1
Phosphorus; total	241.7	69.5	-247.6
Escherichia coli	1,437,568	1,612,196	10.8
Aluminum	19.6	63.1	68.9
Barium	6.04	3.77	-60.2
Beryllium	0.0151	0.0112	-34.8
Calcium	9,510.4	5,300.2	-79.4
Cadmium	0.175	0.364	51.9
Cobalt	0.283	0.709	60.1
Chromium	0.302	0.739	59.1
Copper	15.6	110.8	85.9
Iron	14.3	38.5	62.8
Magnesium	2,329.0	904.1	-157.6
Manganese	0.59	9.09	93.5
Molybdenum	0.45	0.84	46.2
Nickel	0.75	0.78	4.2
Lead	1.62	3.45	53.0
Strontium	36.1	17.5	-106.4
Titanium	0.17	1.40	87.7
Vanadium	0.77	0.94	18.5
Zinc	3.37	11.02	69.5

[a] *E. coli* unit area loads are expressed as colony forming units (CFU) per m^2.

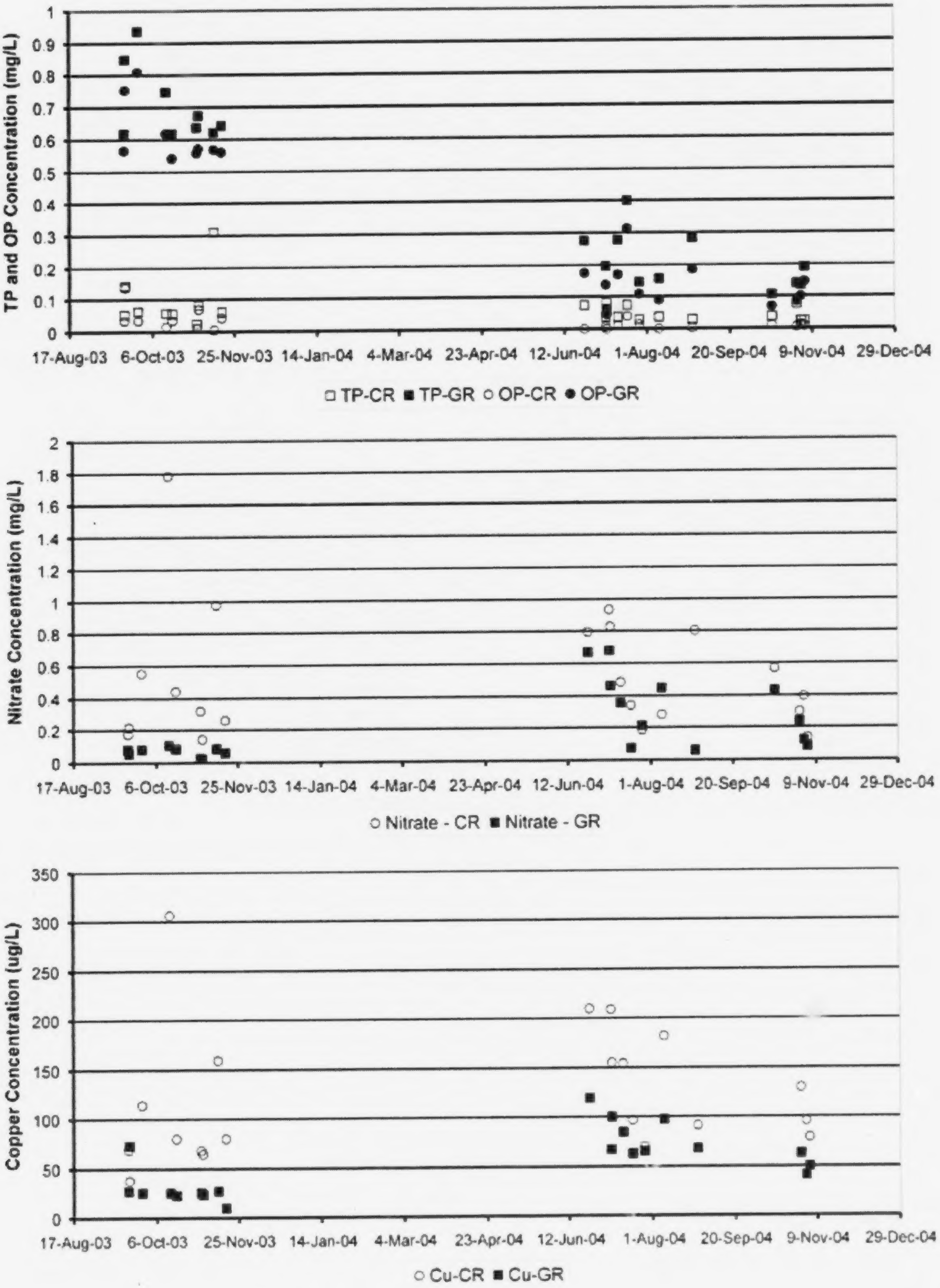

Fig. 5. Green roof (GR) and Conventional roof (CR) concentrations of total phosphorus, orthophosphorus, nitrate, and copper.

TABLE 4. Growing media composition (as provided by manufacturers) and grain size laboratory test results

Growing media	*Composition*	*Grain size test results[a] (%)*				*Comments*
		Gravel	*Sand*	*Silt*	*Clay*	
A	5–10% organic matter and 70% porous mineral aggregate (by volume)	75	21	3	2	Crushed brick, blond peat, perlite, sand and compost from vegetable matter
B	50–60% organic matter, 30% mineral aggregate (by volume)	7	58	17	18	Bark compost, perlite, blond peat, and compost from vegetable matter
C	10–15% organic matter and 55% porous mineral aggregate (by volume)	63	30	4	3	Crushed brick, blond peat, perlite, sand and compost from vegetable matter
D	75% expanded clay and 8% organic matter (by weight)	53	39	4	3	Remaining 17% are materials balanced to the needs of the project. Expanded clay is in the form of gravel sized aggregates.
E	100% expanded clay	97	2	0.6	0.3	Drainage layer used under growing media. Expanded clay is in the form of gravel sized aggregates.
F	25% compost, 25% Solite coarse, 25% perlite, 25% sand (by volume)	77	19	3	1	
G	25% fine compost, 25% Solite fine, 25% perlite, 25% sand (by volume)	20	67	10	3	
H	30% slag, 10% perlite, 30% compost and 30% sand (by volume)	54	40	4	3	
I	25% coarse sand, 25% fine brick, 25% compost and 25% limestone screenings (by volume)	13	77	7	3	
J	Not available	65	6	23	6	
K	Not available	84	13	2	1	
YU[b]	Not available	Not available				Crushed volcanic rock, compost, blonde peat, cooked clay and washed sand

[a] Size classes as follows: gravel: >2mm; sand: 0.05 – 2 mm; silt: 0.002 – 0.05 mm; clay: <0.002 mm.

[b] YU = York University green roof.

and K as per the manufacturers' request. The York University growing media is no longer manufactured, and thus a leachate test was not conducted on this sample. Bulk chemistry analysis of the York University growing media was conducted at the outset of the study.

Grain size analyses showed that 8 of the 11 media tested consisted of more than 50% gravel (>2 mm). These large particles in the media are usually lighter and less dense than typical gravel to ensure that the structural load on the roof is kept to a minimum. Medium E consists primarily of gravel-sized particles because, strictly speaking, it is not a growing medium but rather a drainage layer intended for use in conjunction with one of the other growing media.

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Table 5 presents the results of the bulk media chemistry analyses and typical background concentrations for agricultural soils in Ontario (OMOE 1998). Most constituents in the media were similar to or lower than the background concentrations. The sodium adsorption ratio and conductivity levels were slightly elevated, but not to levels that would adversely affect plant growth. Soil background concentrations were not available for nutrients. However, relative to one another, medium D had the highest phosphorus concentrations (1,469 µg/g), followed by the York University growing medium (1,100 µg/g). The drainage layer, medium E, also displayed high phosphorus levels (960 µg/g). The median phosphorus concentration for substrates A to K was 516 µg/g.

Medium I, with the highest percentage of sand (Table 4), had the lowest phosphorus and nitrogen concentrations overall. This medium was the only one of the 11 with TKN concentrations below 1,000 µg/g. There was an abundance of calcium and iron in the media, but trace metal concentrations were generally low. Medium D contained the highest concentrations of several metals, including barium, chromium, cobalt, copper, and nickel.

Growing media samples were leached four times. The first and fourth samples were analyzed for water chemistry. The pH and turbidty of leachates were higher than observed in the field, probably due to the lower level of dilution in the leachate tests. In the fourth set of leachates, the pH and turbidity of media ranged from 9.1 to 9.6, and 13 to 56 NTU, respectively. The higher turbidity was likely a result of small clay particles less than 1.5 microns, as leachate concentrations of TSS were low (2 to 8 mg/L). The range of leachate alkalinity concentrations varied within the same general range as the field samples (50 to 74 mg $CaCO_3$/L versus 43 to 86 mg $CaCO_3$/L for field samples).

Figure 6 presents leachate results and bulk media concentrations (where available) for nutrients (P and N) and variables with leachate concentrations that frequently exceeded Ontario receiving water standards (OMOE 1994). Exceedances of the standards should not be interpreted as a violation since the standards were not intended for this purpose. Results show a decline in nitrogen and phosphorus concentrations from the first to fourth leachate test for most media. Nitrate concentrations in the fourth leachate were below 0.1 mg/L for all media except medium D. Although total phosphorus levels decreased from a median concentration of 1.2 mg/L in the first leachate to 0.69 mg/L in the fourth leachate, levels remained well above the target of 0.03 mg/L.

The concentration of metals in media leachates varied substantially among growing media (Fig. 6). Copper was leached at relatively low concentrations from 8 of the 11 media tested. Aluminum concentrations were above receiving water standards in several media, and often increased from the first to the fourth leachate. Iron and vanadium levels decreased after the first leachate in most media. Leachate concentrations of lead and zinc were below their respective standards of 5, 1, and 20 µg/L in all media except F (Zn: 92 µg/L; Pb: 6.0 µg/L) and the drainage layer E (Zn: 72 µg/L; Pb: 6.5 µg/L).

Physical properties had little apparent effect on the levels of nutrients or metals in the leachate samples (Table 4 and Fig. 6). One possible exception is the expanded clay drainage layer (medium E), which exhibited lower nutrient leaching and higher concentrations of aluminum, iron, zinc, copper, and lead than most other media. Growing medium B, with 50 to 60% organic matter, had the highest leachate concentrations of phosphorus, but nitrogen levels were relatively low. Media F and G, with 25% compost, had nutrient levels comparable to those observed in other growing media with less compost.

The chemical composition of bulk media was not correlated with the quality of leachates from the same media (Fig. 6). There were several instances in which concentrations were low relative to other media, but leachate concentrations were high (e.g., copper in medium F), and vice versa (e.g., phosphorus in media D and E).

Leachate concentrations of phosphorus (0.22 to 1.63 mg/L) were generally higher than concentrations in runoff from the York University green roof (0.06 to 0.85 mg/L). The green roof also showed lower levels of aluminum, iron, and vanadium than the leachates from the 11 growing media tested, despite generally higher bulk media concentrations. By contrast, total nitrogen levels were lower in the fourth leachate tests (0.04 to 0.83 mg/L) than in the green roof runoff (0.56 to 2.52 mg/L). Copper concentrations in the University green roof runoff were also higher than the leachates, with the exception of the fourth leachate concentrations of media E (118 µg/L) and F (127 µg/L). Copper flashing downstream of the green roof discharge point and upstream of the sampling point may have contributed to elevated copper levels in runoff.

Discussion

By reintroducing natural processes of evapotranspiration and soil water retention into the urban environment, green roofs help to protect streams and replicate the natural hydrologic cycle that existed prior to urbanization. Although green roofs do not infiltrate water or contribute to groundwater recharge, the seasonal pattern of flow volumes closely resembles that to which receiving streams and aquatic life are adapted. This resemblance would be even closer if green roofs were combined on the same site with other stormwater infiltration best management practices, such as permeable pavements, that help to promote groundwater recharge and augment stream baseflows in urban areas.

Runoff results indicated that the green roof retained 63% more rainfall than the conventional roof over the 18 month monitoring period, excluding the winters. Liu and Minor (2005) reported a comparable retention rate of 57% for two green roof plots on a building in Toronto. These two plots were monitored over a similar time period (March to November, 2003 and 2004), but had thinner substrates (7.7 and 10 cm) and a more gentle slope (<2%) compared with the present study. In Ottawa, a green roof with a 15-cm substrate and a 2% slope was found to retain approximately 54% more rain than a similar sized conventional roof (Liu 2003). The Ottawa study included monitoring during the winter (November 2000 to November 2001), when retention of runoff drops to approximately 20 to 30% (Mentens et al. 2006). Together these studies suggest that, for stormwater planning purposes, an annual runoff coefficient of between 0.45 and 0.55 (depending on the slope and substrate depth) should be assigned to extensive green roofs in southern Ontario.

Green roofs help to improve stream water quality

TABLE 5. Chemical characterization of bulk growing media[a]

Variable (units)	Ontario bkg. conc.[b]	MDL[c]	Bulk growing media[d,e]											MDL[f]	York University growing media[g]
			A	B	C	D	E	F	G	H	I	J	K		
GENERAL CHEMISTRY															
Total volatile matter (%)	–	–	8.4	62.0	17.7	10.3	0.2	6.0	6.3	10.0	3.2	7.0	5.3	–	–
Oil & grease (µg/g)	–	20	1,248	5,693	1,913	537	71	1,027	1,128	1,330	615.4	840.8	403.7	–	–
CEC (meq/100g)	–	–	14.9	12.7	14.5	20.3	4.6	41.6	33.8	52.9	23.8	14.7	51	–	–
pH (–)	–	–	7.4	6.0	6.7	6.7	7.3	8.2	7.8	8.0	8.1	7.3	7.9	–	–
SAR (–)	1	–	1.41	1.46	1.59	2.95	0.67	4.79	5.37	3.33	0.51	2.88	1.88	–	–
EC (mS/cm)	0.47	–	1.40	1.75	0.72	1.18	0.21	0.72	0.93	0.40	0.30	0.54	0.03	–	0.48
NUTRIENTS															
Total phosphorus (µg/g)	–	1	489	375	565	1,469	960	639	420	657	239	511	516	20	1,100
Total organic carbon (%)	–	0.1	8.5	35.4	13.7	4.9	0.6	3.4	4.6	5.32	1.69	7.24	3.69	–	–
Total Kjeldahl nitrogen (µg/g)	–	5	1,739	2,986	3,071	2,174	3,324	1,984	1,367	1,731	191	3,196	1,440	10	1,200
METALS															
Antimony (µg/g)	1	1	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	–	–
Arsenic (µg/g)	14	1	3	1	3	4	7	1	2	2	2	4	2	–	–
Barium (µg/g)	190	1	90	111	82	194	182	12	15	137	70	114	85	0	78
Beryllium (µg/g)	1.2	0.5	<MDL	<MDL	<MDL	<MDL	0.66	<MDL	<MDL	0.77	<MDL	<MDL	<MDL	0	0.8
Cadmium (µg/g)	1	1	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	0	0.4
Calcium (µg/g)	–	50	11,323	14,089	13,898	13,384	10,516	10,348	85,304	84,897	82,727	20,874	64,065	0	9,100
Chromium (µg/g)	67	1	10	6	9	87	29	6	6	15	8	15	11	0	31
Cobalt (µg/g)	19	1	5	2	4	14	11	2	3	4	3	7	3	0	12
Copper (µg/g)	56	1	15	17	19	49	24	6	8	26	5	26	11	0	10
Iron (µg/g)	–	10	13,411	4,619	11,439	15,178	21,717	4,418	4,970	7,755	7,493	10,252	8,532	0	19,000
Lead (µg/g)	55	2	7	8	7	<MDL	26	3	4	10	<MDL	20	4	0	1
Manganese (µg/g)	–	1	317	397	336	528	1,182	234	202	1,122	296	404	524	0	300
Mercury (µg/g)	0.16	0.05	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	0.01	0.0005
Molybdenum (µg/g)	2.5	2	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	0	0.25
Nickel (µg/g)	43	2	10	5	9	39	30	6	10	9	6	13	5	0	37
Selenium (µg/g)	1.4	1	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	–	–
Silver (µg/g)	0.35	0.3	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	–	–
Vanadium (µg/g)	91	1	14	9	15	19	41	3	4	11	10	15	14	0	38
Zinc (µg/g)	150	1	76	100	81	75	91	25	27	133	45	68	61	0	31
Chromium (VI) (µg/g)	2.5	1	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	<MDL	–	–

[a] Acronyms and abbreviations: MDL = method detection limit; bkg = background; conc. = concentration; CEC = cation exchange capacity; SAR = sodium adsorption ratio; EC = electrical conductivity.

[b] Typical background concentrations for agricultural soils in Ontario (OMOE 1998).

[c] MDL for samples analyzed by Entech laboratories.

[d] Samples analyzed by Entech laboratories.

[e] Underlined values represent concentrations above background concentrations.

[f] MDL for samples analyzed by Ontario Ministry of the Environment laboratories.

[g] Samples analyzed by Ontario Ministry of the Environment laboratories.

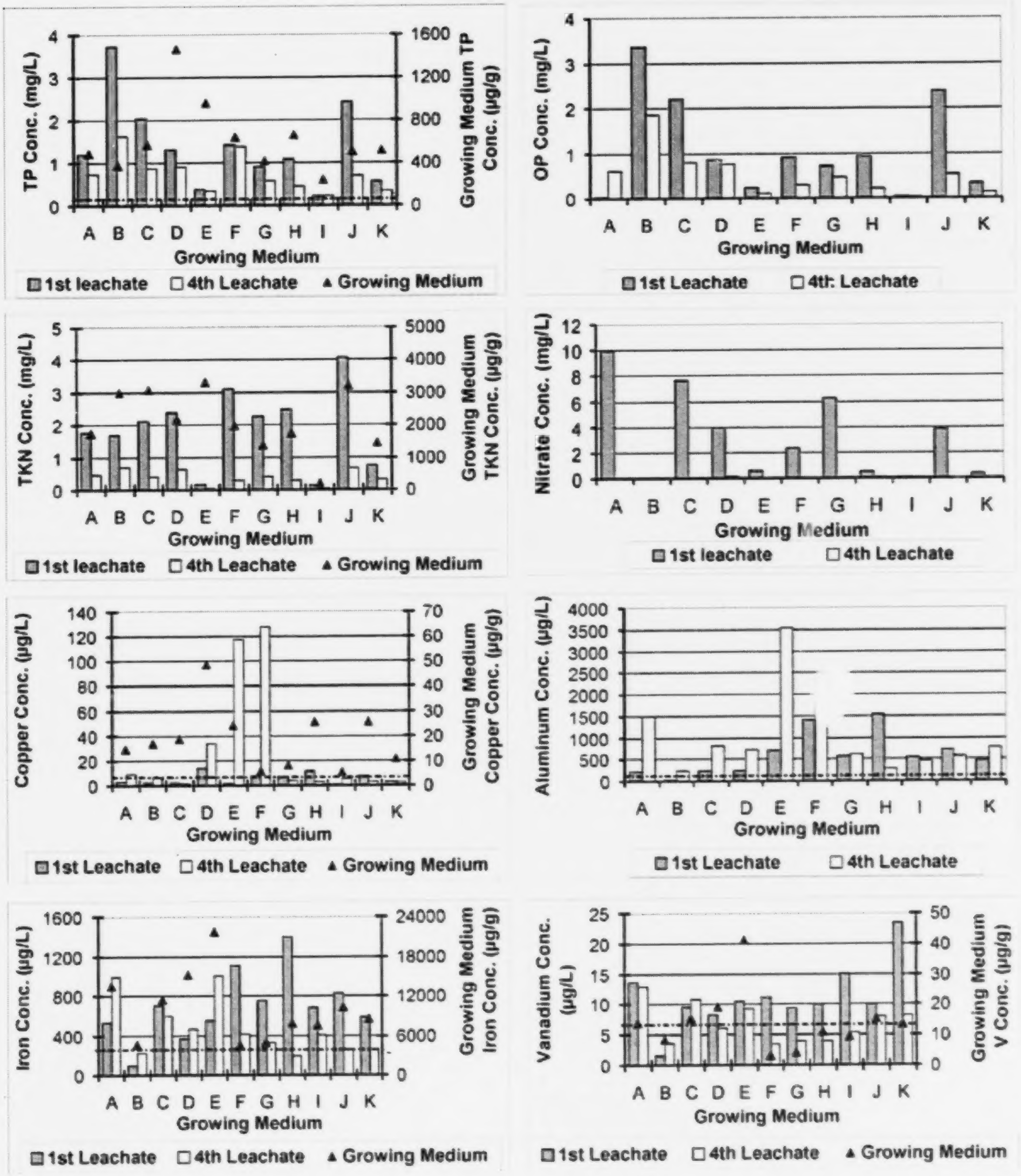

Fig. 6. Leachate and growing media concentrations, where available, for 11 growing media. Note: Black dashed lines represent Ontario receiving water guidelines (OMOE 1994). There are no Ontario guidelines for orthophosphate, nitrate and TKN.

by capturing and retaining atmospheric pollutants, and replacing traditional roofing materials with more natural surfaces. Typical contaminants leached from conventional roof materials include zinc, lead, copper, and PAHs. Coal tars and pitches used in roofing shingles are a source of lead and PAHs (Clark et al. 2001; Van Metre and Mahler 2003). Elevated levels of zinc are common on metal roofs (Van Metre and Mahler 2003). In this study, the conventional asphalt roof was found to contain higher levels of zinc, copper, and PAHs than the green roof, although zinc rarely exceeded receiving water objectives. The metals are of particular concern because, unlike the green roof, the conventional roof had relatively low hardness levels, which increases the potential toxicity of zinc and copper to aquatic life. The ability of green roofs to neutralize acid precipitation and increase water hardness is an important water quality benefit of the technology.

Phosphorus was the main contaminant of concern in green roof runoff, as concentrations were well above the receiving water guideline, and green roof phosphorus loads were considerably higher than the conventional roof. The decline in concentrations over time likely represents a process of leaching whereby soil phosphorus is gradually flushed from the growing medium during the first few years of operation. This would suggest that phosphorus levels may have been even higher during the first year of operation, prior to initiation of monitoring, and that levels may continue to decline in the future, possibly even to concentrations meeting receiving water objectives (0.03 mg/L). As soil phosphorus declines, however, fertilizers may need to be added to support continued plant growth, initiating a renewed cycle of leaching.

The green roof concentrations of total nitrogen (TN: 0.6 to 2.5 mg/L) and total phosphorus (TP: 0.06 to 0.94 mg/L) in this study were lower than reported by Hathaway et al. (2008) in North Carolina, where composted cow manure comprised 15% of the green roof media. In the North Carolina study, TN and TP concentrations ranged from 0.7 and 6.9 mg/L and 0.6 to 1.4 mg/L, respectively. A lower TP range was reported by Hutchison et al. (2003) on two green roofs in Portland (0.2 to 1.1 mg/L). In Pennsylvania, Berghage et al. (2007) found phosphorus concentrations in runoff from three green roofs (mean: 0.28 to 0.58 mg/L) that were more in line with those observed in our study. Nitrate concentrations in the Pennsylvania study were also similar, with the exception of two spikes close to 5.0 mg/L in March (TN was not reported). In Estonia, Teemusk and Mander (2007) reported very moderate levels of TP (0.03 to 0.09 mg/L) in green roof runoff, but TN concentrations (1.0 to 2.1 mg/L) similar to those in our study. The differences among studies may be attributed to a variety of factors, including the composition of growing media, the type and extent of vegetative cover, fertilizer use, and the age of the roofs.

Further testing would be needed to explain the unexpected increase in copper and nitrate concentrations from 2003 to 2004 (Fig. 5). Seasonal effects may have been a factor since the higher concentrations and ranges were generally observed during the months of June, July, and August in 2004, while there were no data for these months in 2003. However, corresponding seasonal changes were not observed for other metals and nitrogen species, as may be expected if season had played a dominant role. Chemical reactions and biologically mediated transformations within the growing media over the winter and spring are other possible causes, but more data would be needed to speculate on the mechanism by which these reactions may affect copper and nitrate concentrations.

There has been some interest in the potential reuse of green roof runoff for toilet flushing, laundry, and irrigation. On the York University green roof, high levels of bacteria and colour would likely prevent such a consideration in the absence of treatment. *E. coli* densities ranged from a low of 4 CFU/100 mL to a high of 5,100 CFU/100 mL, well beyond the acceptable range for reuse, even as irrigation (Exall et al. 2004). While samples were not analyzed for colour, there was a visible tint to all green roof water samples, likely from the fulvic and humic acids in the peat and organic matter. Berghage et al. (2007) reported a similar result in Pennsylvania, indicating a colour range for individual green roof runoff events of roughly 100 to 1,000 PtCo colour equivalents. While the samples from York University had a distinct colour, they were very clear, with turbidity levels below 3 FTU, and TSS less than 5 mg/L.

Green roof growing media are typically composed of light weight materials, such as expanded slate, clay, or perlite with a relatively small amount (<20%) of organic matter (e.g., peat, humus, compost) to provide the chemical properties needed for plant growth. Starters or slow release fertilizers are often added to aid in establishing plants soon after the medium is installed (Beattie and Berghage 2004). The chemical composition of bulk media analyzed in this study generally fell within the range of what may be expected of agricultural soils in Ontario (Table 5). The inclusion of clay minerals rich in aluminum and iron in the growing media can result in high runoff concentrations of these constituents initially (Fig. 6), but field results suggest that concentrations stabilize over time to levels below receiving water standards (Table 2). Leachate concentrations of copper, zinc, and lead were elevated in 3 of the 11 media, suggesting that media may differ substantially in their ability to leach these contaminants. Under field conditions, this study and others (Berndtsson et al. 2006; Berghage et al. 2007) suggest that green roofs may act as a source of some metals, but concentrations are generally much lower than urban runoff, except when drainage water comes into contact with metal roofing and piping materials (Forster and Knoche 1999; Berndtsson et al. 2006).

The absence of any clear relationship between the chemical composition of growing media and leachate concentrations was expected since the potential for leaching will depend on whether constituents are present in the growing media in readily leachable forms. In general, particulate forms that make up part of the mineral substrate would leach less readily than the same constituents present as soluble precipitates. Artificial fertilizers added to promote start-up growth of vegetation would be particularly susceptible to leaching. In Germany and Sweden, the use of coated slow release fertilizers has been shown to help reduce leaching of nutrients (Fischer and Jauch 2002; Emilsson et al. 2007).

Conclusions

The results of this study indicate that vegetated roofs help to mitigate the adverse effects of stormwater on receiving waters by reducing runoff volumes, controlling peak flows, altering the timing of flows, and improving the quality of roof runoff for most constituents. During the growing season, runoff from the green roof was 63% less than from the conventional roof. The capacity of the green roof to retain water combined with more natural vegetated surfaces (relative to traditional roofs) translated into lower loads of most pollutants of concern relative to the conventional roof.

In green roof runoff, phosphorus was the only variable that posed a potential threat to receiving waters. Phosphorus loads from the green roof were considerably greater than from the conventional roof. A significant decrease in phosphorus concentrations from the first to the second year of sampling suggested that phosphorus is leached from the media over time, and that further decreases may be expected as the green roof ages. Nitrate concentrations in runoff were very low and the green roof may even act as a sink for nitrate.

An analysis of the chemical and leachate quality of various growing media available commercially also showed that phosphorus is leached at relatively high levels from most media. Reductions in phosphorus leaching may be achieved by filtering runoff through reactive materials that render phosphorus insoluble by binding it to sediment. Several materials that have shown promise in lakes and coastal areas (e.g., Berg et al. 2002; Robb et al. 2003) require testing in green roof applications. In general, growing media containing phosphorus-rich fertilizers should be avoided.

The laboratory tests indicated that other constituents of concern in green roof runoff may include copper, aluminum, iron, and vanadium, but field results suggest that elevated concentrations of these do not persist over the long term. Copper was the only constituent for which median green roof runoff concentrations exceeded Ontario standards, and in that case roof materials other than the growing medium may have contributed to elevated levels. To help preserve water quality, construction materials and growing media used in green roof installations should be selected to minimize leaching of contaminants into runoff.

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Street Sweeping as a Method of Source Control for Urban Stormwater Pollution

Quintin Rochfort,[1]* Kirsten Exall,[1] Jonathan P'ng,[2] Vicky Shi,[2] Vesna Stevanovic-Briatico,[3]
Sandra Kok,[1] and Jiri Marsalek[1]

[1]*Environment Canada, 867 Lakeshore Road, Burlington, Ontario, Canada L7R 4A6*
[2]*Toronto Water, City of Toronto, 55 John St., Toronto, Ontario, Canada M5V 3C6*
[3]*Transportation Services Division, City of Toronto, 55 John St., Toronto, Ontario, Canada M5V 3C6*

The effectiveness of street sweeping as a source control measure for stormwater pollution was tested at a site in Toronto, using three types of sweepers employed by the City. A paired-plot experimental design was employed along an arterial road with a traffic volume of 26,000 vehicles/day. Typically, after several days of dry weather, one roadway plot was swept by the available sweeper (treated) and the following plot was left unswept (control). After sweeping, sediment on the roadway was sampled on both plots; wet samples were collected by washing off one half of each plot, and dry samples were collected by vacuum cleaning the remaining halves of both plots. Differences between swept and unswept plots were assessed by comparing: (a) conventional sediment quality parameters, total residue mass, and particle sizes for dry sediment samples, and (b) toxicity, conventional water quality parameters, and particle sizes in wet samples. Results were highly variable and contained large uncertainties. The greatest environmental benefits of sweeping were the reduction of the total mass of sediment on road surfaces and a reduction in some dissolved metals in the runoff (e.g., Cr and Zn).

***Key words*: best management practices (BMPs), source control, street sweeping, urban runoff pollution**

Introduction

Modern stormwater best management practices (BMPs) include various types of control measures applied at different spatial scales (MOE 2003). In recent years, much attention has focused on source controls, which represent "practices that prevent pollution by reducing potential pollutants at their source before they come into contact with stormwater..." (WEF and ASCE 1998). One such practice is street sweeping (also called street cleaning in the literature), which has been practiced in urban areas for centuries, but primarily as an aesthetic or housekeeping measure serving to improve street appearance. Only during the last 40 years has street sweeping been considered as a pollution control measure, serving to reduce the mass of pollutants on street surfaces potentially available for wash-off by street runoff (FWPCA 1969). After the FWPCA study, many others followed, but often with inconclusive or contradictory results. In the Nationwide Urban Runoff Program (NURP) (U.S. EPA 1983), no statistically significant reductions in pollution of stormwater from areas swept by conventional (brushing) equipment were found, and Sartor and Gaboury (1984) characterized such equipment as somewhat ineffective in removing the finest particles (<250 µm) carrying a large load of pollutants.

There have been considerable advancements in street sweeping technologies since the NURP studies, with newer street sweepers incorporating vacuum or regenerative air technologies. The vacuum assisted machines draw sediments, dislodged from the road surface by brushes, and deposit them in a hopper. The regenerative air sweepers use a blast of air to dislodge particles from the road surface, collect dislodged particles by vacuum action, and transport them into the separation chamber where filters clean the air prior to discharge or reuse. Advances in sweeping technology then generated the need to revisit the issue of stormwater pollution source control by sweeping.

The literature relevant to street sweeping studies can be divided into the following categories: (a) Accumulation of pollutants on street surface, (b) Efficiency of sweepers in collecting street deposits of solids, (c) Modelling improvements in stormwater quality by street sweeping [i.e., combining studies (a) and (b)], and finally (d) Changes in the quality of stormwater from swept streets or catchments. All of these studies are useful for developing an understanding of street sweeping benefits, or the lack thereof. Solids accumulation on street surfaces is a complex dynamic process, which is affected not only by land use activities, but also by processes reducing those accumulations by wind, traffic generated air currents, rainfall, street sweeping, and material decay (Shivalingaiah 1984). Chemical composition of solids accumulations shows a high pollution potential, with presence of heavy metals, polycyclic aromatic hydrocarbons, nutrients, and chloride (in cold climate areas) (Stone and Marsalek 1996; Deletic and Orr 2005). The efficiency of modern street sweeping equipment in collecting fine materials from the street surface has greatly improved, with the

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* Corresponding author: Quintin.Rochfort@ec.gc.ca

most modern equipment (regenerative air) being capable of collecting "micron-sized" particles, and the overall collection efficiency in controlled (indoor) tests as high as 95% (ETV 2008), and only slightly lower (92%) in the case of freshly dispensed street sweepings in a 1-m swath on a street surface (Breault et al. 2005). Sweeping equipment may also produce some solids by surface attrition, as noted by German and Svensson (2002) in experiments with an industrial brush/vacuum sweeper, when applied on sites with low solids accumulations. At the same time, it is recognized that street sweeping does not remove oil and grease (WEF and ASCE 1998), and the presence of oily films or other processes may turn street solids into cohesive materials (aggregates), the presence of which greatly reduces the sweeping performance in actual conditions. The availability of solids accumulation and characteristics data, "potential" effectiveness of sweeping equipment, and local rainfall frequency data resulted in many modelling studies of street sweeping effectiveness. Using the aforementioned information, one can model the solids accumulation process, with removals caused by rainfall or sweeping, and the modelled pollutant buildup can be used to assess the resulting improvement in stormwater quality (e.g., Taylor and Wong 2002a, 2002b). In general, modelling results indicate potential improvements, which significantly exceed the actual performance observed in field studies.

Actual removals of pollutants from street surfaces very much differ from sweeper performance testing, and are affected by the aforementioned consideration of street sediments, sweeping frequency versus storm interevent times, the type and condition of the sweeping equipment, the operator's training and experience, and obstruction of sweeping operations by parked vehicles (WEF and ASCE 1998). For these reasons, the NURP (U.S. EPA 1983) report suggested that the only way of reliably assessing the environmental benefits of sweeping was by examining stormwater quality at the catchment outfall.

In stormwater quality change studies, only Sutherland and Jelen (1997), who investigated the runoff from loading docks (a different, more controlled situation than streets) concluded that modern sweepers, employing brushing/vacuuming biweekly, would be effective in reducing annual stormwater pollutant loads. Street sweeping appeared to have little effect on stormwater quality as reported in: (a) the NURP program (working with older mechanical sweepers, U.S. EPA 1983), (b) the study of Selbig and Bannerman (2007), even though they noted in their study with brush/vacuum sweepers a "reduced presence of road deposited sediments" after sweeping, and (c) a study of highway sweeping, in which brush/vacuum equipment malfunctions and poor pavement conditions led to high result variability (Waschbusch 2003).

Thus, the literature on the environmental effectiveness of street sweeping offers diverse results, with a general conclusion that aesthetic benefits and some improvements in stormwater quality can be achieved by using the most advanced well-maintained sweepers at optimized cleaning frequencies in areas with high accumulation of street debris (WEF and ASCE 1998). Furthermore, before adopting new, more intense street sweeping programs, one also needs to consider the associated costs and undertake a benefit/costs analysis.

Recognizing this often conflicting information on street sweeping benefits in the literature, and the proposed use of street sweeping as a stormwater pollution source control measure in the Toronto Wet Weather Flow Management Master Plan (City of Toronto 2003), the City of Toronto partnered with the National Water Research Institute to study environmental benefits of street sweeping with respect to: (a) reductions in sediment deposits on street surfaces due to sweeping with various types of sweepers but without comparisons of performance of individual sweepers, and (b) reductions in constituent concentrations (and toxicity) of street wash off.

Study Area

For the purposes of this study, an experimental area, defined as a section of roadway without interfering driveways and intersections, was selected on Markham Road just north of McNicoll Avenue, in Toronto, Ontario, Canada (Fig. 1). This road is located in a commercial / industrial area and comprises three traffic lanes in each direction, with a total traffic volume of 26,000 vehicles/day. Subsections of the curb lanes on both sides of the road, delineated by a single catchbasin drainage area (approximately 50 m in length), were marked out for "test" (swept) and "reference" (unswept) experimental plots. Swept areas were immediately followed by unswept areas (in the direction of travel), on both the northbound and southbound lanes. Two pairs of plots (four plots) were investigated each time: northbound swept (NBSW), northbound unswept (NBUS), southbound swept (SBSW) and southbound unswept (SBUS). The paired reference site was used for comparisons against the swept (treated) sites to gauge the effectiveness of the sweeping process, since the road dust in the unswept areas could be different on each side of the road, depending on the traffic flow and local air transport. Within each of these four sites, the pavement area was subdivided into "wet" and "dry" sampling sections; wet sampling occurred over the area immediately up-slope from the catchbasin and dry sampling occurred further away (see Fig. 1).

Methods

Street Sweeping

Three types of street sweepers were available during various phases of the study. During the 2004 field season, sweeping was done with an old-technology regenerative air (ORA) sweeper and a conventional mechanical sweeper (CM). The ORA sweeper employs a turbine fan

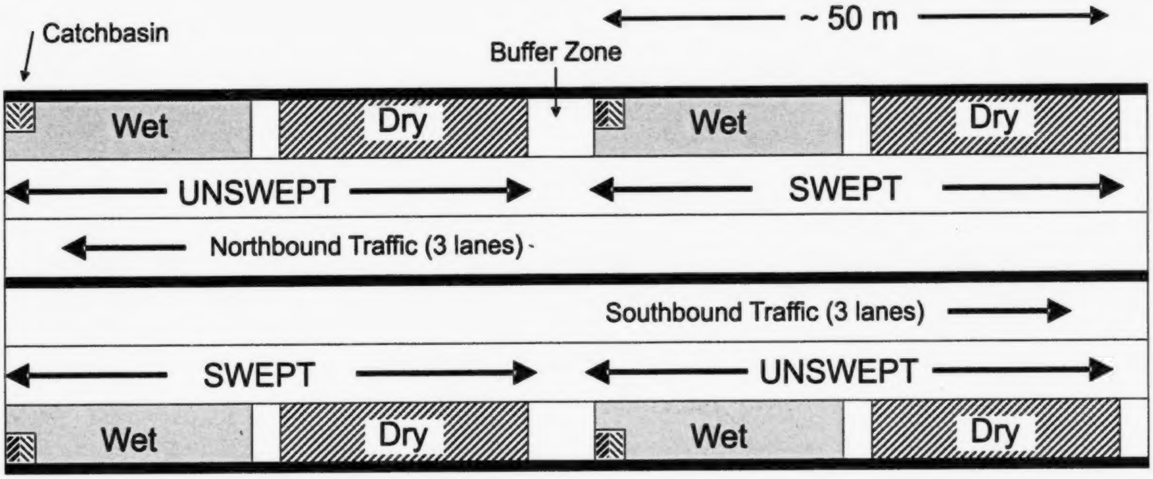

Fig. 1. Diagram of field site sampling locations.

and pickup head to apply a vacuum suction to the road surface and lift the sweepings into the hopper by this vacuum action. Gutter brooms with steel tines remove compacted debris from the curbs into the pickup head path. The CM sweeper uses a large counter-rotating main broom and gutter brooms to sweep road debris into a conveyer system which transports debris into the hopper.

In the 2005 and 2006 field seasons, two models of the new-technology regenerative air (NRA) sweepers were used. Both employ a technology which reuses air in a closed loop system that blasts air under pressure to dislodge sweepings from the street surface across the pickup head path and transfers them pneumatically into a collection hopper. Air containing the fine road dust (particulate matter) is cleaned by filtration, which prevents the fine road dust from being released into the ambient air environment. Gutter brooms move compacted debris from the curbs to the pickup head path. In 2005, an older model (not maintained by the City) was used, but in 2006 a new City-customized model was made available for this study.

All of the tested sweepers were equipped with dust control sprays, but these were not used during these tests in order to maintain dry surfaces for sampling. The operational speed of the sweepers was between 5 and 15 km/h, depending upon the model. Street sweepers were applied only to the "swept" experimental plots; the reference plots remained unswept. Sampling occurred between June and November each year. Following sweeping operation, dry and wet sampling was performed at each test plot. No routine sweeping took place at this site during test periods so that the solids could build up on the road surface during periods of dry weather. Ideally, sampling would take place after seven days or more of dry weather, but this was not always possible. The antecedent dry period (defined as the number of consecutive days before sampling with <7 mm of rainfall) ranged between 1 and 20 days, with an average of 7 days.

Dry Sampling

Only the curb lanes were sampled at these sites, since previous studies have shown that almost all road-deposited sediment and litter accumulate within 1 m of the curb (Sartor and Boyd 1972; Novotny 2003). An area of asphalt pavement and concrete curb, 20 m in length by 4 m in width (the width of the curb lane), which was furthest away from the test catchbasin, was brushed and vacuumed with a powerful industrial vacuum (Nilfisk-Advance 2050, constructed of stainless steel), to collect a sample of the road-deposited sediment. The broom was moved back and forth gently in front of the vacuum head to loosen any attached fine particles. The vacuum head was moved from the curb to the road crown in overlapping strokes; each pass was overlapped by one-half the width of the vacuum head each time. The vacuum head was also run along the concrete curb edge to collect material retained in the corners. Total sampling time was 20 minutes, allowing the operator to pace the collection of the sample and maintain the same technique in each case.

Samples were removed from the vacuum using stainless steel trowels and disposable brushes, and were stored in amber glass jars. Total mass of the sample was determined for each site prior to sample splitting. Samples were split using the coning and quartering method, and a whole subsample was prepared for particle size analysis. The rest of the sample was sieved using two mesh sizes to create three fractions: <64 µm, 64 µm to 2,000 µm, and >2,000 µm. Sample fractions >2,000 µm were weighed and discarded. The remaining two fractions were submitted to a Canadian Association for Environmental Analytical Laboratories (CAEAL, now CALA) accredited laboratory for chemical analyses using in-house analytical methods based either on the U.S. Environmental Protection Agency methods (U.S. EPA 2008) or *Standard Methods* (APHA et al. 2005). Polycyclic aromatic hydrocarbons (PAHs) were Soxhlet extracted using a hexane / acetone mixture, and the samples were run in hexane on a gas chromatograph equipped with a mass-selective detector (GC-MSD) operating in selective ion mode (SIM) (based on U.S. EPA 2008). All of the remaining solids-based analytical methods were based on *Standard Methods* (APHA et al. 2005). Acid extractable metals were determined using a microwave-assisted, concentrated nitric acid extraction of the sediments, followed by inductively coupled plasma-mass spectrometry (ICP-MS) analysis. Total Kjeldahl nitrogen (TKN) and total phosphorus (TP) were digested using sulfuric acid (with potassium sulfate and mercuric oxide catalysts) at 380°C, and analyzed using an automated spectrophotometer. Total organic carbon (TOC) was analyzed by high temperature combustion with nondispersive infrared (NDIR) detection.

Wet Sampling

The wet sampling was performed on an area equal (in length) to that used for the dry sampling (20 m by 4 m), but located immediately upstream of, and directly draining into, the catchbasin. The round catchbasin grate (D = 0.61 m) was removed and replaced with a catchbasin insert that collected all of the runoff from the site. The insert was sealed to the catchbasin with a flexible temporary caulking and a small recirculating pump was lowered to the bottom to provide mixing and facilitate sample removal from the insert. In addition, berms were placed downstream of the catchbasin to ensure that all of the runoff was retained within the test area. Wash water was municipal tap water which had been dechlorinated by bubbling air through it and by further aging the water overnight. A stainless-steel pump and reservoir fed either a gentle rain-like spray head (used during the first year) or a water broom (more focused water jets) which was used to generate runoff. The surface was washed from the crown of the road towards the curb and then down along the curb. For most tests, 110 L of tap water was applied within the 16 minutes and generated roughly 70 L of runoff. This simulated a rainfall event with an intensity of 5.16 mm/h, and a runoff coefficient of 0.64, indicating the initial hydrological abstraction (surface wetting and depression storage) of 0.5 mm.

Three 20-L pails of wash-off water were collected for toxicity testing. Laboratories at the Ontario Ministry of Environment performed both rainbow trout (*Oncorhynchus mykiss*) LC50 96-hour static nonrenewal tests (EPS 1/RM/13, Environment Canada 2000a) and *Daphnia magna* LC50 96-hour static nonrenewal tests (EPS 1/RM/11, Environment Canada 2000b). Microtox tests were performed at Environment Canada. The acute Microtox EC50 15-minute test was performed on the samples following the standard protocol (AZUR Environmental 1998).

Water samples collected at the sites were analyzed for total suspended solids (TSS) according to *Standard Methods* (APHA et al. 2005). Particle size analysis was performed using a Malvern laser particle size analyzer. All other sample analyses were performed by a CAEAL-accredited laboratory. PAHs were liquid/liquid extracted using dichloromethane and analyzed by gas chromatography-mass spectrometry in selective ion mode (SIM) (U.S. EPA 2008). All remaining analyses were performed using methods based on *Standard Methods* (APHA et al. 2005). Total and dissolved (0.45-µm-filtered) metals were analyzed by ICP-MS. TKN and TP were digested using sulfuric acid at 380°C and analyzed using an automated spectrophotometer. TOC was analyzed by high temperature combustion with nondispersive infrared detection.

All data collected from both wet and dry sampling were analyzed using a statistical software package (Graphpad / Prism V. 4.03). Box and whisker plots were used in preference to parametric statistical tests based on a normal distribution of data and requiring a higher number of replicates. Log-transformed data sets were analyzed using the Wilcoxon Matched-Pairs Signed-Rank test with $\alpha = 0.05$, in order to determine the significance of differences between median values before and after sweeping.

Results and Discussion

Road Deposited Sediment

Road deposited sediments (RDS) were defined as those substances originating from the erosion of surrounding soils, dry and wet atmospheric deposition, biological inputs from leaf material, road surface wear, road marking degradation, vehicle wear (tires, brakes, body, etc.), vehicle fluids, and particulate emissions (Pitt and Clark 2003). In spite of the relative wealth of literature on street sweeping (reviewed, e.g., in Pitt et al. 2004), the knowledge of street sediment processes and of their variability, including sediment removal by washing/sweeping and rainfall/runoff, appears to be incomplete. Sediment accumulation rates greatly varied at the plots studied, ranging from 18 to 405 kg/curb km in individual runs, for comparable dry weather periods of accumulation, but from different sides of the road. In these accumulations, the finest fraction (silt and clay) represents a minor component, typically less than 6% of the total accumulation; the rest is gravel and sand. Recognizing that all types of sweepers should easily remove coarse particles, field comparisons of sweepers suffer from low sensitivity of such tests; differences in removing the finest fraction may be masked (overshadowed) by experimental errors and low numbers of field tests (given by frequent rainfalls). For this reason, no comparisons of the four different sweepers employed at various times in this study have been attempted. In fact, for low accumulations (about 30 to 50 kg/curb km), it was impossible to discern between RDS accumulations in unswept and swept plots.

Figure 2A shows the results of sediment loading analysis before and after sweeping for all runs combined

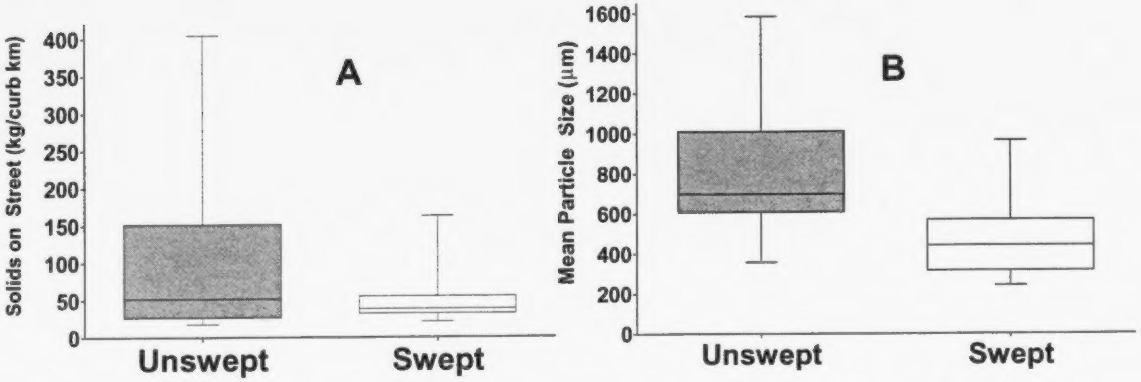

Fig. 2. A. Total mass of solids (RDS) collected at swept and unswept sites ($n = 25$ data pairs); **B.** Particle size of RDS at swept and unswept sites ($n = 25$ data pairs).

($n = 25$). It was noted that there were large differences in RDS mass between the southbound and northbound lanes, which contributed to the large difference in upper and lower limits in the figure. The difference in the median values of RDS mass before and after sweeping was found to be statistically significant ($p = 0.0236$) according to the Wilcoxon Signed-Rank test. Additionally, sweeping reduced variability in the mass of solids on the road under various conditions, as indicated by the reduction in span between upper and lower limits after sweeping (i.e., the whiskers), as well as the smaller interquartile range (i.e., the box).

The difference between the southbound and northbound lanes was further analyzed by plotting the total RDS mass versus the number of dry days (Rochfort et al. 2007). While the northbound data showed a typical trend—slowly increasing rates during the first 1-3 days, followed by more or less constant accumulation values (30-50 kg/curb km) for longer dry weather periods—the southbound data showed a number of high rates (160-360 kg/curb km) greatly exceeding the normal rates, which were reported in the literature to range from 50 to 115 kg/curb km (Pitt et al. 2004). Accumulations of RDS observed at the study site greatly varied, both temporally and spatially, and no clear correlation with the antecedent dry period could be detected.

The very high rates indicate the presence of another mechanism contributing to RDS build up, which could be construction traffic through the site, with loaded trucks travelling southbound and empty northbound. Particle size analysis showed that the composition of RDS samples was highly variable, with gravel content ranging from 4 to 42% by weight; sand from 54 to 91%; and silt and clay from 2 to 6%. Mean particle size of the sediment samples from unswept sites was 544 μm northbound and 761 μm southbound; both sizes correspond to a coarse sand classification.

With respect to sweeping data, there appeared to be a lower RDS loading limit of about 30 kg/curb km, below which sweeping was ineffective (compared to the 'no sweeping' alternative). Within the realm of experimental uncertainties, for high RDS loadings, sweeper effectiveness was largely due to better removal of gravel (up to 88%) and sand (up to 62%).

These findings are confirmed by other studies. In six trials performed by Waschbusch (2003) with the Enviro Whirl street sweeper (which employs brushing and vacuum action), the sweeper was capable of removing 50% of the RDS at a loading of 140 kg/curb km (the upper limit for their tests); for comparison, the NRA sweeper employed in our study removed 65% at the same loading rate (which was mid-range for our tests). Waschbusch (2003) experienced the same lower limit of RDS of 28 kg/curb km (100 lb/curb mile), below which sweeping becomes ineffective in RDS removal. Recognizing the cost of sweeping, it appears that sweeping operations are not beneficial in areas with low RDS mass, or when such accumulations were reduced by an antecedent rain storm.

Following particle deposition on street surfaces, other processes take place: particle aggregation, consolidation, coating by oil, decay (Shivalingaiah 1984), mechanical processing by sweeping equipment (brushing, wetting, vacuuming), and removal by rainfall and runoff. These processes are poorly understood and this contributes to large uncertainties when assessing the effectiveness of street sweeping in pollution control. Notwithstanding this caveat, the data collected by dry sampling of experimental plots in our study indicate that the sweepers produced significant reductions ($p < 0.05$) in the case of relatively large solids accumulations. It can be speculated that these sweepers would provide some measure of stormwater pollution source control if applied, according to the recommendations of WEF and ASCE (1998) in urban areas where high solid deposits exist (e.g., major traffic corridors), at a relatively high sweeping frequency (related to the local rainfall inter-event time), with well-trained staff using well-maintained sweeping equipment (which should be kept in peak operating condition to maximize the efficiency of sweeping).

TABLE 1. Qualitative reductions of solids and chemical loads (mass/curb km) from roads by sweeping[a,b]

	Solids	*TOC*	*TKN*	*TP*	*Cu*	*Cr*	*Pb*	*Zn*	*Phe*	*Fl*	*Py*
Dry RDS	– –	δ–	δ+	– –	δ–	δ–	δ–	δ–	δ+	δ+	δ+
Road wash-off	δ–	δ+	δ–	δ+	δ–	– –	++	– –	δ+	δ+	δ+

[a] Significance of differences evaluated by Wilcoxon signed-rank test with $\alpha = 0.05$: δ– statistically insignificant reduction; – – statistically significant reduction; δ+ statistically insignificant increase; + + statistically significant increase.

[b] Phe = phenanthrene; Fl = fluoranthene; Py = pyrene.

The mean particle sizes on individual swept and unswept plots were calculated by the method of moments, which produces a weighted average based on the entire particle size distribution (Folk 1965). Sweeping significantly ($p < 0.0001$) reduced the median value of the above mean particle sizes in RDS on swept plots by 37%, indicating a preferential removal of coarser particles (Fig. 2B). It is important to note that even for the best performing sweeper, there were solids left on the road surface after sweeping that had a mean particle size of over 400 µm, and the mass of fine particles (<64 µm) left on the surface was over 2 kg/curb km.

Removal of RDS from street surfaces is beneficial for such reasons as reducing the entry of sediment into runoff and sewers, and reducing export of sediments from the catchment and their deposition in downstream BMP structures, with concomitant adverse impacts of such deposits on BMP structure performance (e.g., reductions in the dynamic storage of stormwater management ponds). Concerning stormwater quality, the main interest is in removing chemical constituents from the street surface, which after rainfall may contribute to either suspended or dissolved loads of pollutants in stormwater. Qualitative removals of eleven constituents (solids, TOC, TP, TKN, Cu, Cr, Pb, Zn, and the PAHs phenanthrene [Phe], fluoranthene [Fl], and pyrene [Py]) by sweeping street solids are summarized in Table 1. The data in the table indicate that there were only 2 statistically significant removals (solids with $p = 0.023$ and TP with $p = 0.006$), and the remaining parameters displayed no statistically significant change.

Two examples of constituent removals are shown in Figs. 3A and 3B for TP and Zn, respectively. The TP data closely mimic the solids removal chart (Fig. 2A), similarly showing a significant ($p = 0.027$) reduction after sweeping, and indicating that TP is mostly bound to solids. Zinc was removed in experiments with high solids loadings (>50 kg/curb km), which was also true for all the other trace metals and PAHs studied. In general, however, sweeping did not significantly reduce the mass of Zn in RDS ($p = 0.214$).

Concentrations of chemicals in RDS are also of interest, partly because of concerns about disposal / reuse of street sweepings, and partly because such information contributes to the wealth of data on pollutant accumulations on streets. Detailed listings of such concentrations can be found in Rochfort et al. (2007); median concentrations (in mg/kg) of the metals Cr, Cu, Pb, and Zn, and the PAHs Phe, Fl, and Py in whole RDS samples (i.e., <2,000 µm) are presented in Table 2. The concentrations of metals in RDS actually increased slightly with sweeping, as would be expected when larger solids (i.e., those with lower relative surface area and lower metal concentrations) are preferentially removed.

Similar findings were reported by Deletic and Orr (2005), and Robertson and Taylor (2007) who monitored three urban sites in Manchester, U.K. Lead and Zn displayed local variability in time, whereas Fe and Mn showed temporal variability, which was consistent across sites. As expected, the silt and clay fraction (<63 µm) contained the highest Pb, Cu, and Zn concentrations,

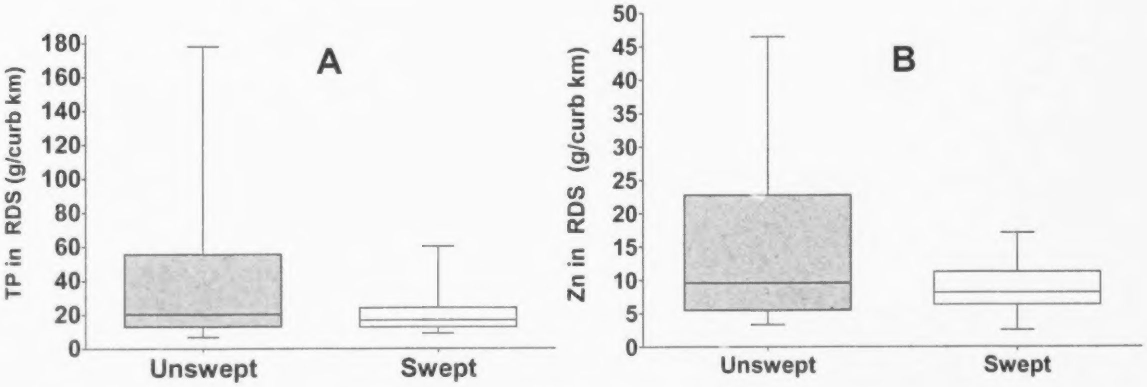

Fig. 3. A. Mass of TP in RDS ($n = 25$); **B.** Mass of Zn in RDS ($n = 25$).

TABLE 2. Selected median trace metal and PAH concentrations in street sediment (<2,000 μm) from swept and unswept test sites compared with sediment quality guidelines

	Sample size "n"		*Median metal concentration in RDS (mg/kg)*				*Median PAH [a] concentration in RDS (μg/kg)*		
			Cr	*Cu*	*Pb*	*Zn*	*Phe*	*Fl*	*Py*
Sediment quality guideline		ISQG [b]	37.3	35.7	35	123	41.9	111	53
		PEL [b]	90	197	91	271	515	2,355	875
Unswept	25		50.5	76.9	27.0	170	258	452	347
Swept	25		70.3	81.2	37.7	196	398	666	561

[a] Phe = phenanthrene; Fl = fluoranthene; Py = pyrene.

[b] ISQG = interim sediment quality guideline; PEL = Probable Effect Level. Source: Canadian sediment quality guidelines (CCME 2002)

which would cause the greatest environmental concerns due to risk of resuspension and inhalation, and transfer to receiving waters during rain events. Furthermore, these fine particulates may not be easily controlled by street sweeping.

The median concentrations of all of the selected metals and PAHs exceeded the interim sediment quality guideline (ISQG; CCME 2002), but not the probable effect level (PEL), values in almost all cases. The only exception was that of Pb in the unswept samples, which was slightly below the ISQG value. In 50% of all RDS samples from unswept sites, one or more PEL guideline limits for the selected metals or PAHs were exceeded, and such materials would represent a pollution risk if transported to the receiving waters. A comparison of the PAH concentrations of RDS samples from this study with PAH concentrations from a multilane divided highway (Marsalek et al. 1997), found that those in the current study were 10 times lower. As sweeping primarily removes coarser particles with lower concentrations of trace metals and PAHs, the concentrations of trace metals and PAHs in RDS samples from swept sites showed increasing trends, but few of these comparisons were statistically significant.

Furthermore, concentrations of Cr, Pb, and Zn compared well with those reported by Stone and Marsalek (1996) for an industrial city, but concentrations of Cu, Pb, and Zn represented only 20 to 50% of those in sediment from the aforementioned highway, with a 3.6-times higher volume of traffic (Marsalek et al. 1997). Note also that metal concentrations in northbound lane RDS were consistently greater than those in the southbound lane RDS, supporting the notion of southbound RDS dilution by construction truck traffic.

Road Wash-Off

Comparisons of dry and wet sampling of experimental plots in this study revealed differences in sizes of the particles collected by these two methods. Catchment wash-off in wet sampling removed practically only clay and silt from the street surface (more than 90% of the total mass was in particles <50 μm), with a median particle size of approximately 10 μm. The total mass removed by this process greatly exceeded the mass of silt and clay removed by dry sampling with gentle brushing and vacuuming with an industrial vacuum cleaner. This suggests that washing may be a more effective process for detaching the finest particles from the street surface or breaking up fine particle aggregates than brushing/vacuuming, as also observed by Deletic and Orr (2005). In a few cases, increased solids wash-off was noted after sweeping. This was also reported in earlier studies by other researchers who surmised that those solids left behind after sweeping became detached from the surface and were fine enough that they could then be more easily washed off (Sartor and Gaboury 1984), or that the solids removed provided a protective covering for the fine material, which was then exposed and thus more easily washed off the surface. It was also noted by German and Svensson (2002) that sweeping "produced" some solids by attrition of road surfaces.

Median particle size in street wash-off collected in the catchbasin was 10.1 μm for the unswept areas, and 9.59 μm for the swept areas, indicating that there was little to no effect of sweeping on the size of particles transported by "gentle" washing from the street surface to the sewer inlet. Overall, the median for the total solids mass washed off from unswept sites was 9.8 kg/curb km; or just 19% of the total RDS mass collected by vacuuming. Assuming the same loading of both (wet and dry) plots, this showed that only a small portion of the material that the sweepers left on the street was actually transported by wash-off during a simulated rainfall event; the actual amount washed off during rainfall would vary depending upon the intensity and duration of the rainfall. In the wash-off experiments, the equivalent rainfall intensity was low and duration short. Furthermore, the kinetic energy of rain drops landing on the street surface may be important for dislodging particles from the surface, and this process was not simulated in wet sample collection.

It has been suggested that the RDS possesses high contents of organic materials, and the surface of coarser sediment particles is coated with various chemicals, oil, or fine particles (Brinkmann et al. 1999). As the sediment ages on the road surface, it may become cohesive (or

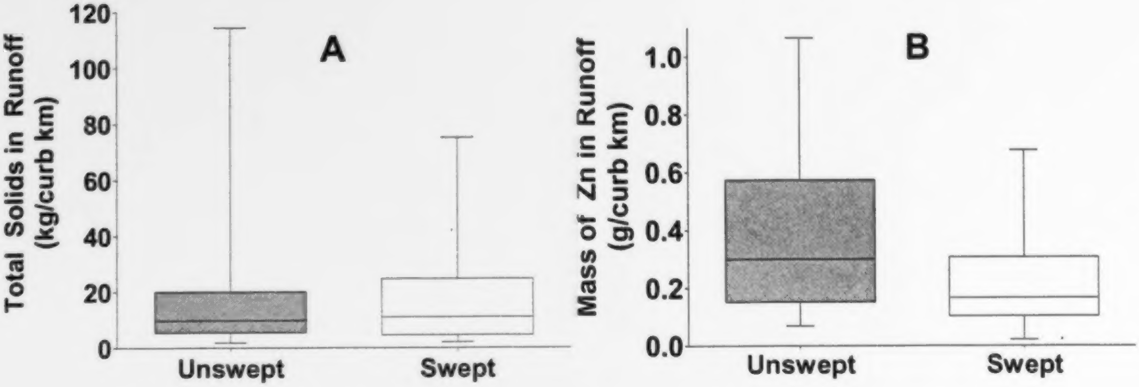

Fig. 4. A. Changes in total solids washed off ($n = 25$); **B.** Changes in dissolved Zn washed off ($n = 25$).

aggregated) and more resistant to wash off. Street sweeping may remove this coating and make the fine particles susceptible to wash off. This theory was not supported by observations in the current study. Figure 4A shows that with minor exceptions (i.e., when there were very high masses of solids on unswept sites), the mass of solids washed off during wet sampling was relatively unaffected by sweeping. The Wilcoxon Signed-Rank test confirmed that differences in the masses of solids washed off of swept and unswept areas were statistically insignificant ($p = 0.224$). High (dry) RDS loads in 2006 contributed to the highest observed values of solids washed off; solids levels were lower in 2004 and 2005. The change in 2006 may have been the result of alterations to the type of sediments being imported into the area. On average, washing removed only 19% of the RDS on unswept plots, and 29% on swept plots. Further attention has turned to dissolved constituents, summarized in Table 1, to determine whether dissolution could be a more effective transport mechanism than sediment transport.

Statistically significant reductions in constituent transport were only observed for dissolved Zn ($p = 0.0001$) and Cr ($p = 0.0005$), indicating reduced runoff of these constituents from swept plots (e.g., Zn shown in Fig. 4B). Surprisingly, dissolved Pb in runoff was significantly ($p = 0.002$) increased after sweeping, indicating increased susceptibility to wash-off after sweeping or some contribution of Pb by the sweepers themselves. Dissolved Cu, TOC, TKN, TP, and the PAHs Phe, Fl, and Py in wash water were not significantly affected by sweeping at the $\alpha = 0.05$ level, although on at least one occasion, the old regenerative air sweeper did appear to release some PAHs during sweeping (this may have been due to transport of PAH contaminated dirt from outside sources into the test area). Transfer of solids from other locations was found to be a potential problem in the operation of sweepers, as also encountered by other researchers (e.g., Waschbusch 2003). It was likely that the November 10, 2004 sweeping operation had a much higher level of contamination than on other test dates since the same street sweeper was also being used in nearby residential areas to clear up leaf debris at that time. This additional material was likely the source of solids contamination for the runoff TSS and street dry sampling. The recommendation for future testing was to pre-clean the sweeper and then provide a "clean" area to run the sweeper as a warm-up loop, to ensure that contamination would be minimized. Sampling in 2005 and 2006 suggested that the transfer of solids from outside sources was no longer a problem during the tests.

Irish et al. (1998) found that TSS in stormwater runoff could be impacted by street sweeping and the antecedent dry period. The intensity of the previous storm event was also a factor in predicting TSS in runoff. Street sweeping would therefore be an important means of controlling TSS in runoff. Constituents such as oil and grease, chemical oxygen demand, and biochemical oxygen demand were more likely to be predicted by traffic intensity during a storm event. Nutrients such as TP and TKN were best predicted based on traffic counts during the antecedent dry period and the total duration of the dry period. Metals such as Cu and Pb were predicted by traffic volume during a storm event, and street sweeping was unlikely to reduce these concentrations. Iron was better predicted by dry weather period, but street sweeping did not appear to influence the concentrations in runoff. Zinc in runoff could be mostly determined from traffic volume during the antecedent dry period and the intensity of the previous storm, and they concluded that other stormwater management BMPs might be better than street sweeping to control this metal. In this study, however, Zn was found to be removed from runoff when using the regenerative air sweepers.

When assessing the sweeping efficiency from runoff data, the high variability of runoff and sediment quality data is a major concern in these types of studies. Kang and Stenstrom (2008) reanalyzed a number of previous studies to determine if enough samples had been collected to provide the statistical power required to detect

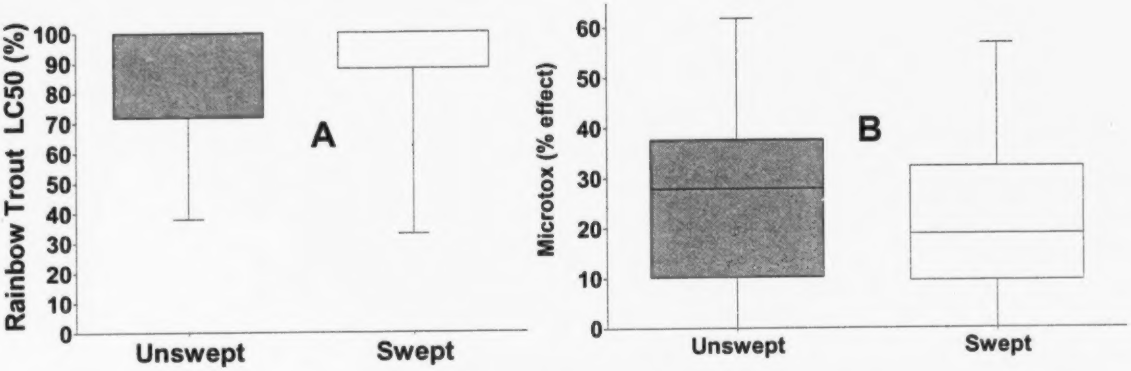

Fig. 5. A. Changes in toxicity to rainbow trout (note: 100% is nontoxic) ($n = 25$) **B.** Changes in Microtox 15-minute % effect (note: % effect is the reduction in light output and therefore a greater % effect is more toxic) ($n = 25$).

significant differences. They found that only four studies had enough replicate results to determine significant differences in runoff characteristics. Of those four studies, only one showed a significant result: a reduction in concentration of TSS in runoff after sweeping (Irish et al. 1998). Kang and Stenstrom (2008) also noted that street sweeping effects may be difficult to detect in runoff due to the variability of the antecedent dry period, rainfall intensities, and the transport of particulate material by wind and other processes. They suggested that street sweeping may reduce part of both the airborne and the wash-off components of the RDS. This makes it more difficult to attribute differences in runoff quality directly to street sweeping practices.

Road Wash-Off Toxicity

Wash-off toxicity was evaluated by three acute toxicity tests which had been shown in past studies (Marsalek et al. 1999) to provide a wide range of ecotoxicological responses to urban pollution. Of the three applied tests, the 96-hour rainbow trout LC50 test was found to be the most sensitive at detecting acute impacts in this study; Microtox and *Daphina magna* tests did not show acute toxicity. Figure 5A shows the impact that street sweeping had on the toxicity of the wash-off. The toxicity was defined by the lethal concentration required to kill 50% of the population and was expressed as LC50. When the LC50 was 100% (or greater), it was an indication of no toxicity. As the LC50 value decreased, the severity of the toxicity increased. Rainbow trout toxicity results were highly variable for all sites. During the 2005 testing, very little toxicity was encountered (almost all sites were nontoxic), but in other years toxicity was minor. For comparison purposes, a "fail" in this test was considered to be LC50 <50%. None of the tests showed this acute level of toxicity. This may have been due to a combination of considerable rainfall (and therefore a lack of dry days over which to accumulate pollutants) during the test periods and a lack of heavy traffic. Although on occasion some paired tests showed a slight reduction of toxicity after sweeping, none of the tests demonstrated a clear tendency to reduce the toxicity of the wash-off. It should be further noted that as applied in this study, wash-off with a low volume of water represented the first flush, which is a phenomenon describing high concentrations of pollutants during the initial phase of runoff. Thus, samples that would be collected during later phases of runoff should be even less toxic than those presented here (Marsalek et al. 1999).

Microtox test results did not show any acute toxicity, since the sublethal percent effect result (which measures the reduction in light output by the test organisms) was used to gauge the differences between swept and unswept sites. On this scale, zero percent effect would indicate no impact (no reduction in light output relative to controls); as the percent effect climbs, so does the toxicity. Fig. 5B shows that the effect of street sweeping was to reduce the level of toxicity somewhat, but the median percent effects before and after sweeping were not significantly different ($p = 0.24$). The most toxic samples were found in the southbound unswept runoff in 2006, but the corresponding swept site runoff showed a similar level of toxicity.

The toxicity of samples in this study were low, but given that the 26,000 vehicles per day traffic volume was below the 30,000 vehicles per day threshold proposed by Driscoll et al. (1990) for acute effects to appear, this result may not be unusual. It is also possible that inert sediment from construction areas was responsible for "diluting" toxic sediment generated from traffic. Higher than normal rainfall during the test period may also have resulted in the wash-off of some of the more toxic materials from the test areas prior to sampling. The complete absence of toxicity to *Daphnia magna* was unusual, since in previous studies, *Daphnia magna* reacted quite strongly to urban runoff pollution (Marsalek et al. 1999). The 96-hour rainbow trout LC50 and Microtox EC50 15-minute tests (using the sublethal "percent effect" rather than EC50 values), appeared to be the most sensitive tests to apply at this location in terms of identifying changes in the toxicity of runoff. Even so, only limited differences were discernable at best. Swept sites were generally less toxic than the unswept sites, although since the variation in the results was so great, none of the results were significant at the $\alpha = 0.05$ level. Tiefenthaler et al. (2001) were able to use sea urchin fertilization, mysid survival and growth, and Microtox in toxicity identification evaluations to demonstrate that Zn was a major factor in runoff toxicity for a commercial parking lot, but due to low toxicity at this Toronto site, such determinations were not possible.

Conclusions

Conflicting literature data on street sweeping efficiency and continuing advances in sweeping technology provided the impetus for studying sweeping as a source control measure for pollution conveyed by urban stormwater. The study adopted a new approach to assessing the environmental benefits of sweeping; both toxicity and chemistry were used to evaluate the changes in runoff from swept (treated) and unswept (control) test areas. Within the limitations of this study and the realm of experimental uncertainties, the study results indicate that sweeping reduced significantly street sediment deposits on experimental plots with high sediment accumulations (well in excess of 50 kg/curb km). Such reductions were also reflected in reductions of chemical loads associated with such sediment, particularly with respect to TOC, TKN, TP, Pb, Zn, phenanthrene, fluoranthene, and pyrene. Wet sampling of swept and unswept plots by washing produced somewhat different results; washing produced much higher loads of silt and clay than dry vacuuming, and a significant reduction in washed-off load was only observed for dissolved Zn and Cr. Toxicity results for wash-off from swept and unswept plots were about the same, and indicated no toxicity for *Daphnia magna*, and only limited toxicity for rainbow trout and Microtox. In general, the swept sites were usually slightly less toxic than the unswept ones (not statistically significant).

The experience reported in the literature and gained in this study indicates that the most modern and efficient sweeping operations do provide environmental benefits when applied in areas with high solids deposits. Experimental field proofs of such benefits remain elusive because of extreme uncertainties in the assessment of improvements in stormwater quality due to street sweeping. Furthermore, pollution source control by sweeping should be subject to cost-benefit analysis and, with respect to the protection of receiving waters, compared with other BMPs providing comparable benefits.

Acknowledgments

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Characteristics of Sediment Removal in Two Types of Permeable Pavement

Chris Brown,[1] Angus Chu,[1] Bert van Duin,[1,2] and Caterina Valeo[1]*

[1]*Civil Engineering, Schulich School of Engineering, University of Calgary 2500 University Drive NW, Calgary, Alberta, Canada T2N 1N4*

[2]*Wardrop Engineering Inc. - A Tetra Tech Company, Calgary, Alberta (formerly Westhoff Engineering Resources Inc.)*

This study investigates the processes and characteristics of solids removal in two types of permeable pavement: UNI Eco-Stone and porous asphalt. The mechanisms and processes behind solids removal within permeable pavement structures was studied for these two types of permeable pavements using both field installations and laboratory experiments. Results from the study showed that both pavement types are capable of excellent total suspended solids removal, in the range of 90 to 96% removal of solids from influent. Particle size distribution analysis of accumulated sediment within the pavement structure and in the influent and effluent showed that the particles in the effluent of the pavements are substantially finer than that in the influent. Laboratory results involving no crust formation indicated that, although solids removal occurs throughout the entire structure, the "sieving action" occurs primarily at the geotextile interface.

***Key words*: permeable pavement, total suspended solids, best management practices, particle size distributions, porous pavement**

Introduction

The use of permeable pavements as a method for improving the management and treatment of stormwater has increased dramatically over the past decade. The potential to reduce or eliminate the need for traditional stormwater infrastructure by use of low impact development (LID) technologies has become an important area of research and discussion within the stormwater management profession. Designers, city planners, and stormwater management engineers across Europe, North America, and Australia have begun exploring the use of this best management practice (BMP) as a promising, cost-effective way to address the problem of increased stormwater runoff and decreased stream water quality associated with urbanization (Brattebo and Booth 2003). Because of their ability to allow water to quickly infiltrate through the surface, permeable pavements allow for reductions in runoff volumes and peak runoff rates, as well as improvements in the water quality of stormwater runoff (Collins et al. 2006).

Stormwater passes through the pervious surface of the permeable pavement, and percolates through the layers of the substructure, where it is temporarily stored. In regions with soils that are highly permeable, the stormwater can be allowed to slowly infiltrate into the native soil (subgrade) underneath the permeable pavement (Collins et al. 2006), whereas in areas with low-permeability soils, effluent water can be conveyed underneath the pavement through an underdrain to a storm sewer system (James and Langsdorff 2003).

In this paper, the term "permeable pavement" will refer to the entire permeable pavement structure, rather than just the surfacing material (also referred to as the surface course). Several different types of permeable pavement surface courses exist, each with distinguishing physical characteristics. Two such surface courses, and the ones presented in this study, are porous asphalt and open-jointed paving blocks.

Porous asphalt pavement was first developed in the early 1970s and consists of standard bituminous asphalt in which the fines have been screened and reduced, thus creating small voids and allowing water to pass through (Cahill Associates Inc. 2005). Porous asphalt is usually placed directly on a gravel base course.

Open-jointed paving blocks consist of interlocking load-bearing units (often constructed of concrete) that are shaped such that, when laid, they produce open voids between adjacent units. The voids can then be filled with a porous aggregate or turf (Cahill Associates Inc. 2005). It is this fill material that gives the pavement its porosity and permeability (Ferguson 2006). Open-jointed paving blocks are also referred to as permeable unit pavers, modular interlocking concrete blocks with external drainage cells, or permeable interlocking concrete pavers.

Total suspended solids (TSS) is an important pollutant to be removed from stormwater runoff because it is harmful to aquatic ecosystems. It increases water turbidity, inhibits plant growth and diversity, affects river biota, and reduces the overall number of aquatic species. The specific size of solids present is also an important factor, as it may have different impacts on the above mentioned harmful effects. In addition, several pollutants, including certain heavy metals, lead, phosphorus, and

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* Corresponding author: valeo@ucalgary.ca

polycyclic aromatic hydrocarbons tend to be associated with particulate matter, especially finer particles (Balades et al. 1995; Colandini et al. 1995; Legret and Colandini 1999; Environment Australia 2002; Walker and Hurl 2002; Teng and Sansalone 2004). There is currently debate as to what particular size fraction of TSS is the most important to remove from stormwater runoff, but current City of Calgary and Alberta Environment regulations state that 85% of TSS greater than or equal to 75 microns must be removed on an annual basis (Alberta Environment 2001).

The primary mechanism behind TSS removal is sedimentation and mechanical filtration through the pavement structure (Stotz and Krauth 1994; Urban Water Resources Centre 2002a). Most solids accumulation (and thus removal) has been found to occur in the top several centimetres of the pavement structure (Balades et al. 1995; Pratt et al. 1995; James and Gerrits 2003; James 2004). The geotextile layer above the base course may also play a role in limiting the transport of pollutants into the substructure, and can affect overall filtration efficiency (Pratt et al. 1990; Bond et al. 1999). Effluent from permeable pavements generally has a finer gradation than the influent due to the filtration processes (Legret et al. 1996).

TSS removal efficiency for permeable pavement is highly dependent on the size of particles applied to the pavement surface. Since it acts as a filter, a larger proportion of coarse material will be retained in comparison to fine material. Little research has been done into the analysis of the removal of specific size fractions of TSS by permeable pavements, but it is obviously important to consider the particle size distribution of both influent and effluent when investigating the pavements' capabilities for removing solid matter. Furthermore, as stated previously, the current guidelines for the City of Calgary stipulate a removal rate corresponding to particles over 75 µm through stormwater treatment methods. In recent years there has been a debate in the region over the current pollutant loading criteria, and changes in regulations will likely occur in the future. This further increases the need for reliable regional data regarding the capability of permeable pavements for removing various size ranges of particles; these data can then be used to determine the acceptability of these BMPs for whatever future regulations may stipulate.

Therefore, the objectives of this research were to determine the performance of two types of permeable pavements, specifically porous asphalt and open-jointed paving blocks with respect to water quality and fractions of TSS removal. This will help gain a better understanding of the mechanisms and processes behind solids removal within permeable pavement structures. The methodology to do this employed both laboratory models and field installations within an urban development site in the City of Calgary. A discussion of the impacts of clogging on the long-term infiltration capacity of these two types of permeable pavement is described elsewhere (van Duin et al. 2008).

Methodology

Field Experiments

Installations. Two pilot-scale permeable pavement installations, with surface courses of porous asphalt and UNI Eco-Stone open-jointed paving blocks, were installed in November 2005 on Hochwald Avenue SW in Calgary, Alberta, Canada (see Fig. 1). The street is a collector road located at Currie Barracks, a former military base, and is used both by local traffic and as a shortcut by residents on either side of Currie Barracks. The site is located directly in front of a stop sign, and the road can be categorized as a low volume road that receives fewer than 1,000 vehicles per day including occasional heavy-duty (i.e., Mack truck) vehicles.

Both pavement sections measure approximately 8-m long by 6-m wide, and are on a 3% longitudinal slope. The porous asphalt surface consists of a 65-mm thick layer of open grade friction course with a maximum aggregate size of 11 to 16 mm and no particles finer than

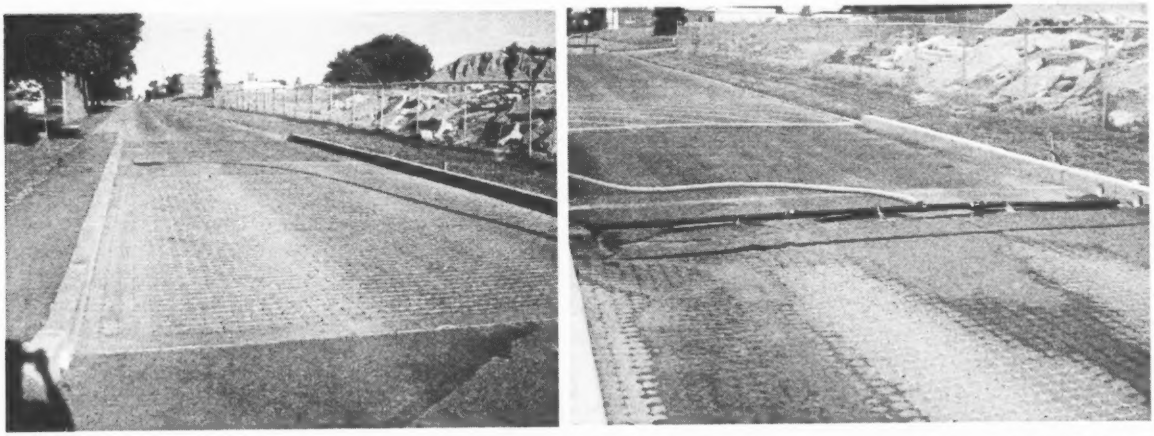

Fig. 1. Currie Barracks permeable pavement installations with field application device shown in right figure (UNI Eco-Stone in foreground, porous asphalt in background).

600 μm. The asphalt has an in-place void space of 18 to 22% and polymer modified binders and fibers. The UNI Eco-Stone blocks measure 115 by 215 mm, and are 80-mm high, with an overall surface void ratio of 12.18%. Both installations contained a 30-mm bedding course layer beneath the surface course, and a 400- to 500-mm base course layer below which was the existing subgrade. Figure 2 details the original design specifications for the Eco-Stone installation, while Fig. 3 shows the gradations of the bedding course layer and base course layer. A Carthage Mills 15% monofilament woven geotextile separated the bedding course and base course layers, and a nonwoven geotextile separated the base course from the subgrade. A perforated pipe underdrain was situated at the bottom of the base course of both pavements, draining to a nearby monitoring manhole.

At the downstream end of each pavement surface was a concrete pad, running the width of the road, which was used during experiments to seal a wooden beam to the surface in order to replicate ponding water scenarios and to ensure no volume was lost from the experiments.

During the experiments, water was pumped from two water storage tanks to a field runoff applicator (shown in the right image of Fig. 1) fitted with five 2-inch (5-cm) outlets which allowed the water to be dispersed evenly across the width of the pavement. The monitoring equipment located in the monitoring manhole consisted of a V-notch weir, a Hach Sigma 900 autosampler, a Hach Sigma 950 flow meter, and a Hach Sigma 75 KHz ultra sonic flow sensor, which measured the level behind the weir.

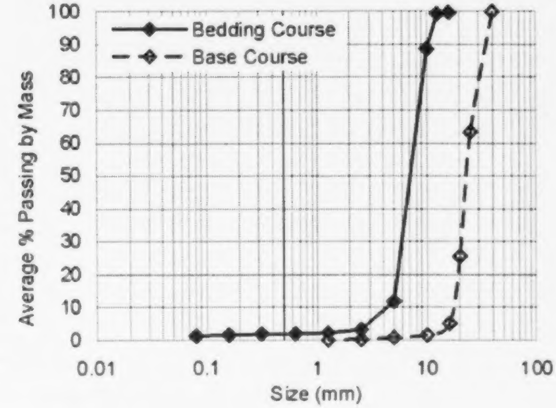

Fig. 3. Gradation analysis of bedding course and base course for Eco-Stone and porous asphalt (and joint fill material for Eco-Stone).

Procedures. The experiments at Currie Barracks consisted of the application of a known volume of water (8,400 L) to the pavement surfaces at a constant flow rate and with constant TSS concentration (500 mg/L). This concentration is close to the average event mean concentration recorded by the City of Calgary over multiple storm events. The application water was stormwater hauled from the 69th Street SW storm pond in order to replicate the biological and chemical properties of stormwater runoff, which may play a role in solids and pollutant removal processes

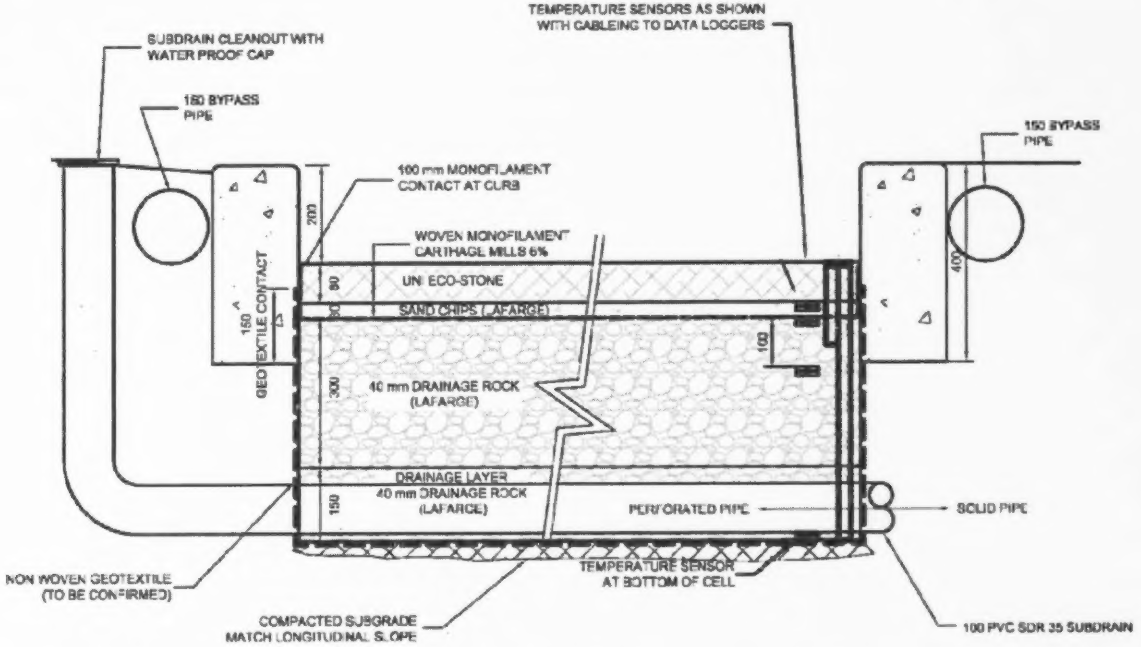
SUBDRAIN CLEANOUT WITH WATER PROOF CAP
150 BYPASS PIPE
100 mm MONOFILAMENT CONTACT AT CURB
WOVEN MONOFILAMENT CARTHAGE MILLS 6%
UNI ECO-STONE
SAND CHIPS (LAFARGE)
40 mm DRAINAGE ROCK (LAFARGE)
DRAINAGE LAYER
40 mm DRAINAGE ROCK (LAFARGE)
GEOTEXTILE CONTACT
200
150
80
30
300
150
100
400
TEMPERATURE SENSORS AS SHOWN WITH CABLEING TO DATA LOGGERS
150 BYPASS PIPE
PERFORATED PIPE
SOLID PIPE
NON WOVEN GEOTEXTILE (TO BE CONFIRMED)
COMPACTED SUBGRADE MATCH LONGITUDINAL SLOPE
TEMPERATURE SENSOR AT BOTTOM OF CELL
100 PVC SDR 35 SUBDRAIN

Fig. 2. Currie Barracks permeable pavement cross-section (Eco-Stone installation shown. Porous asphalt installation is identical except for the surface course.) (Courtesy of Westhoff Engineering Resources, Inc.)

in permeable pavements (Shackel and Pearson 2003). Frequent analysis of the raw stormwater showed the background TSS concentrations averaging between 0 and 10 mg/L; the overall target concentration of 500 mg/L was not significantly impacted.

Water was pumped from the storage tanks to the pavement surface, at an average flow rate of 5.87 L/s, which is equivalent to 242 L/s/ha based on a 48.48 m^2 pavement surface area and *I/P* ratio of 4. The *I/P* ratio is the impervious to pervious area ratio (dimensionless) for the catchment that the permeable pavement services (Urban Water Resources Centre 2002b). In addition to incident rainfall, permeable pavements may receive runoff from adjacent areas. Thus, consideration must be given to the drainage area upstream of the permeable pavement area. Permeable pavements installed in locations with higher *I/P* ratios will naturally receive more runoff and thus require higher maintenance than those installed in locations with lower *I/P* ratios. This factor is often overlooked in the literature when evaluating the performance of permeable pavements in reducing surface runoff. For this study, an *I/P* ratio of 4 was chosen. This is based on the types of situations in which these pavements could be installed, incorporating the findings from long-term plugging and hydraulic performance research (Urban Water Resources Centre 2002b). The equivalent rainfall intensity for the average 5.87 L/s flow rate is about 95 mm/hr. This is slightly greater than the 1-in-100 year storm event value of approximately 80 mm/hr for a 20 minute duration, which is close to the average application time of 23 minutes (The City of Calgary Stormwater Management & Design Manual 2000). Effluent samples were collected with the Sigma 900 autosampler and, along with manually collected influent samples, were analyzed for TSS and particle size distribution.

Laboratory Models

Figure 4 shows two lab-scale pavement models used for "long-term" laboratory experimentation of both pavement types. The structures had cross sectional areas of 465 mm by 465 mm. The layer depths of the pavements for the laboratory models were almost identical to the layers at the field installations with the exception that the base course depth was reduced to 300 mm in order to minimize the overall weight of the model. There was no geotextile below the base course since effluent collected in an underdrain in a field installation would be above this geotextile. Base effluent readings taken directly after construction of the pavement models, that is, prior to the application of the runoff, amounted to 0 mg/L, verifying the absence of any fines in the aggregate.

Runoff applicators located above the pavement models consisted of an array of 16 vinyl distribution tubes connected to a sheet located approximately 10 mm above the pavement surfaces. Water was fed into the applicator from a mixing tank which had four mixing blades controlled by a variable speed DC motor capable of keeping large particles (up to 1 mm) in suspension.

Experiments and procedures. For this series of experiments, volumes of water with specific concentrations of suspended solids were applied to the pavement models for a simulated period of time. Grab samples for the influent and effluent were frequently taken for TSS and particle size distribution analysis. The following relationship was used to calculate the necessary applied volume for an equivalent simulated year:

$$V = dA(I/P + 1) \tag{1}$$

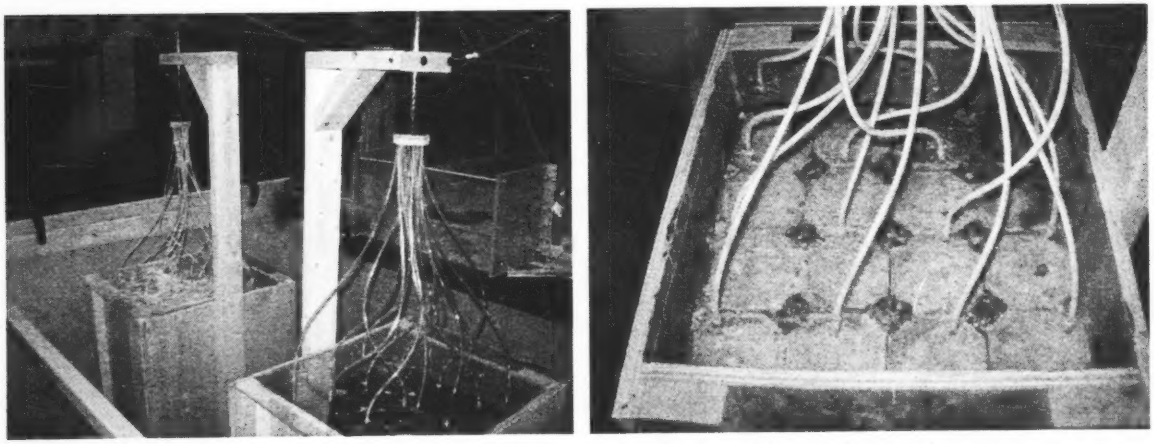

Fig. 4. Laboratory model shown on the left and runoff model detail shown right.

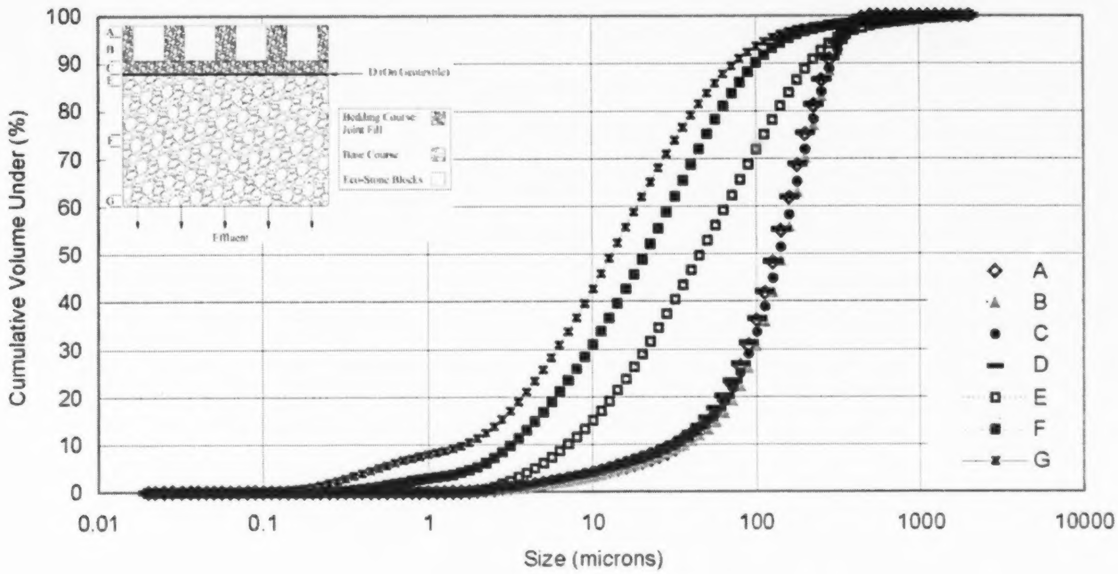

Fig. 5. Particle size distributions for Eco-Stone layers throughout pavement model after simulation of 20 years of runoff (layer locations shown upper left corner).

where V is the equivalent volume of annual precipitation (m^3), d is the average annual depth of precipitation in Calgary (m), A is the permeable pavement area (m^2); and I/P is the impervious to pervious area ratio (dimensionless).

For the pavement models, the total surface area was 0.216 m^2, the average annual precipitation for Calgary was approximately 400 mm, and an I/P ratio of 4 was assumed. Therefore, the application of approximately 0.432 m^3, or 432 L of water (rounded down to 425 L for sake of simplicity), loaded with 500 mg/L of sediment, was required to simulate one year of stormwater runoff for each laboratory model.

Water from the mixing tank was applied continuously to the runoff applicators for both pavement models at the same flow rate of 0.009 L/s. A total of 20 years' runoff and sediment application was simulated for both pavement types to replicate the long-term performance of the two pavement structures. Upon completion of the 20-year simulated application, the pavement models were carefully disassembled, and samples of accumulated sediment were collected from various layers of both pavement models so that the particle size distribution could be analyzed throughout the structures.

For the Eco-Stone model, sediment was collected from the top 25 mm of joint fill material (Location A as shown in insert of Fig. 5); the bottom 55 mm of joint fill material (Location B); the bedding course (Location C); on top of the geotextile (Location D); and three separate depths of the base course (Locations E, F, and G). For the porous asphalt model, samples were collected from the asphalt surface itself (Location A as shown in insert of Fig. 6); the bedding course (Location B); on top of the geotextile (Location C); and from three separate depths of the base course (Locations D, E, and F).

Sediment Characteristics

Experiments were performed with sediment collected from Calgary roads in order to observe realistic long-term performance of the pavement structures. Spring cleaning street sweepings from the City of Calgary were collected and sieved to a size of 250 μm and below.

The sediment was sieved to 250 μm because this represents the finer portion of material that would typically be found on Calgary streets. The finer portion was desired since this is typically the material that is ultimately responsible for the clogging of permeable pavements (Gerrits 2001). With regards to filtration capabilities, it is presumed that if the pavements are capable of removing the sub-250-μm particles, they would be capable of removing larger particles as well. It is also the size range that is of particular importance in stormwater treatment because of its association with particular pollutants and the relative ease with which it can be mobilized and transported into stormwater discharges (Barnes et al. 2001). Research has shown that finer particulates (<250 μm) are more efficient in the adsorption of pollutants, including phosphates, heavy metals, and pesticides, and as such will carry a higher pollutant concentration (Sartor et al. 1974; Roger et al. 1998; Andral 1999).

For both field and laboratory experiments, the same 500 mg/L concentration of sediment was applied to the

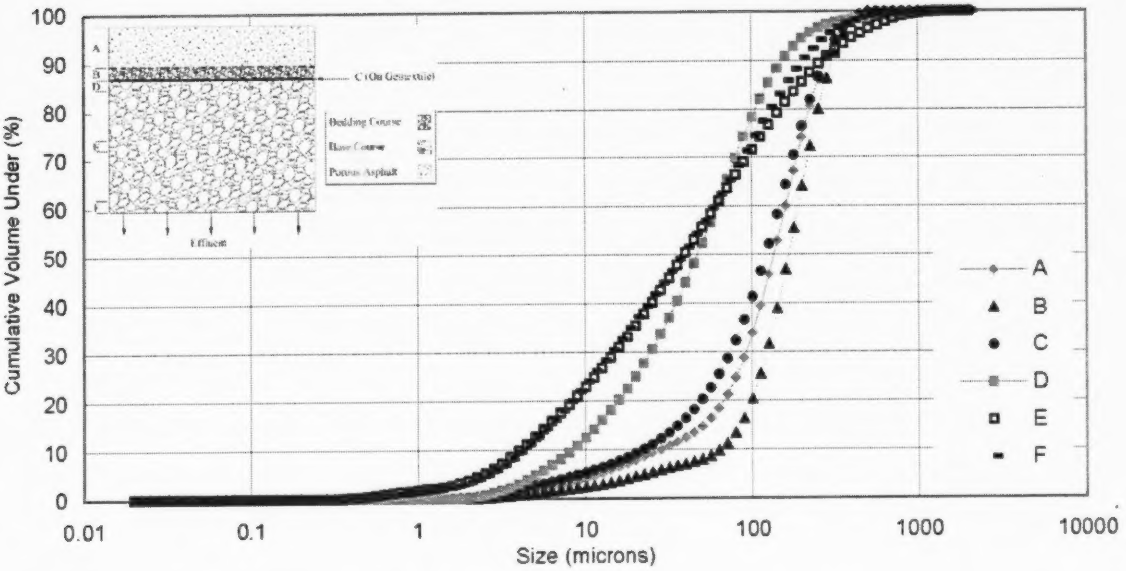

Fig. 6. Particle size distributions for porous asphalt layers throughout pavement model after simulation of 20 years of runoff (layer locations shown upper left corner).

surfaces, and grab samples of inflow concentration were taken consistently throughout all experimental runs. Due to turbulence within the mixing tank, the instantaneous TSS concentration of the inflow varied considerably. However, the overall average over the course of experiments remained 500 mg/L.

For the field experiments, the sediment was applied manually to the pavement surface by, as evenly as possible, dispersing 500 g of sediment over the pavement for every 1,000 L of continuously applied water from the storage tanks through the field runoff applicator. The sediment was applied in this manner because it proved too difficult to keep the sediment in suspension in the field experiments. There can be inconsistencies in the instantaneous concentrations of the applied water, however, given that the experiments at Currie Barracks represent ponding situations, the average concentration of water infiltrating the pavements, or the event mean concentration, would have been very close to 500 mg/L. The influent particle size distributions (PSDs) were taken by collecting 2-L samples at the outlets of the conveyance tubes, centrifuging, and collecting the sediment.

The washed bedding course and base course materials did not contain any particles finer than 500 µm, and the applied sediment was all passed through a 250-µm sieve. Consequently, the sediment at the various layers of the pavement structures was obtained by gently washing the aggregate from each location through a 500-µm sieve and collecting the wash water and sediment from beneath the sieve. All accumulated street sweepings sediment passed through the sieve, while any material from the courses within the pavement structures themselves was retained on the sieve. The collected samples were then centrifuged, separated, and analyzed.

Analytical Methods for TSS

TSS was analyzed according to the procedures described in *Standard Methods for the Examination of Water and Wastewater* (APHA et al. 1999). Particle size analysis was performed with a Malvern Mastersizer 2000 laser diffraction particle size analyzer. A bulk sample of approximately 0.5 to 1 gram of sediment was required for an accurate analysis using the laser diffraction analyzer. To collect this mass of sediment, a considerable volume of effluent was required, in some cases more than 10 L. This liquid sample was centrifuged at 4,000 rpm for 15 minutes in four 1-L bottles per cycle. After removal from the centrifuge, the liquid supernatant was poured off, and the sludge was removed using a spoon. Once the sediment had been collected, it was dried and separated into the appropriate target mass of 0.1 to 0.3 grams. This 0.1 to 0.3 gram subsample was then soaked overnight in hydrogen peroxide (concentration of 30%) to dissolve organic bonds and allow better separation of particles. This naturally dissolves a portion of the organic content, but for low-organic-content samples such as those used in this study, removal of the organic content was not believed to have a significant impact on the particle size distribution. Following this, the sample was soaked overnight in Calgon to properly disperse the particles, at which point it was ready for analysis in the Malvern Mastersizer 2000.

Results and Discussion

TSS Removal Characteristics Observed in the Field Experiments

Both pavement surface types displayed very similar TSS removal efficiencies, with 92 to 98% of applied TSS removed from the simulated runoff influent. Figure 7 shows the particle size distributions of the influent and effluent at Currie Barracks for the Eco-Stone and porous asphalt pavements. These are averages of the results for all the dates when simulated runoff experiments were performed. For the Eco-Stone, the maximum and minimum standard deviations of the influent results for all the run dates and all particle sizes were 8.6 and 1.2%, respectively. For the Eco-Stone effluent, the maximum and minimum standard deviations were 11.0 and 4.3%, respectively. The corresponding maximum and minimum standard deviations for the porous asphalt influent and effluent were 4.7 and 1.1%, and 7.1 and 3.0%, respectively. Substantially finer solids were observed in the effluent as compared with the influent. The effluent PSD did not appear to show any consistent patterns of change over the course of a single experiment, or from experiment to experiment.

Based on average particle size distribution data for all experimental dates, and overall average TSS removal rates for a particular pavement, one can arrive at approximations of removal efficiency for various size fractions within the pavement. The formulas used to calculate removal efficiency of particular size fractions are as follows:

$$\begin{aligned}\beta_{\le x} &= \frac{TSS_{in(\le x)} - TSS_{eff(\le x)}}{TSS_{in(\le x)}}\\ &= \frac{TSS_{in}\alpha_{in} - TSS_{eff}\alpha_{eff}}{TSS_{in}\alpha_{in}}\\ &= \frac{TSS_{in}\alpha_{in} - (1-\xi)TSS_{in}\alpha_{eff}}{TSS_{in}\alpha_{in}}\\ &= \frac{\alpha_{in} - \left[(1-\xi)\alpha_{eff}\right]}{\alpha_{in}}\end{aligned} \tag{2}$$

where $\beta_{\le x}$ is the removal efficiency for all particles less than or equal to X μm; $TSS_{in(\le x)}$ is the TSS in the influent less than or equal to X μm (mg/L); $TSS_{eff(\le x)}$ is the TSS in the effluent less than or equal to X μm (mg/L); TSS_{in} is the TSS in the influent (total, mg/L); α_{in} is the proportion of influent that is less than or equal to X μm; TSS_{eff} is the TSS in the effluent (total, mg/L); α_{eff} is the proportion of effluent that is less than or equal to X μm; and ξ is the average overall removal efficiency of all TSS. Similarly,

$$\begin{aligned}\beta_{\ge x} &= \frac{TSS_{in(\ge x)} - TSS_{eff(\ge x)}}{TSS_{in(\ge x)}}\\ &= \frac{TSS_{in}(1-\alpha_{in}) - TSS_{eff}(1-\alpha_{eff})}{TSS_{in}(1-\alpha_{in})}\\ &= \frac{TSS_{in}(1-\alpha_{in}) - (1-\xi)TSS_{in}(1-\alpha_{eff})}{TSS_{in}(1-\alpha_{in})}\\ &= \frac{(1-\alpha_{in}) - \left[(1-\xi)(1-\alpha_{eff})\right]}{(1-\alpha_{in})}\end{aligned} \tag{3}$$

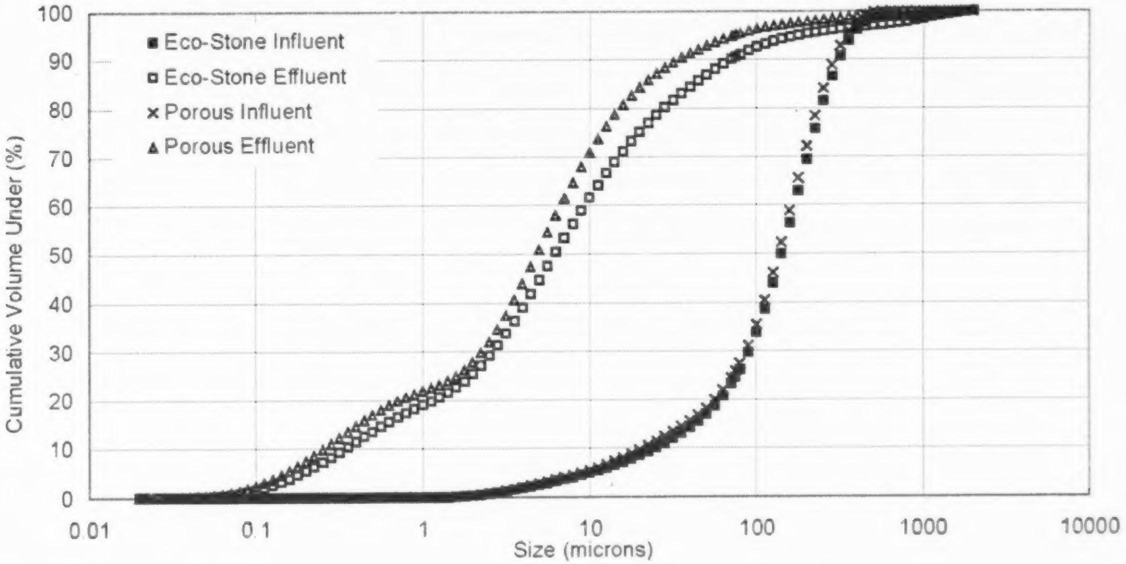

Fig. 7. Average particle size distributions at Currie Barracks.

where $\beta_{\geq x}$ is the removal efficiency for all particles greater than or equal to X µm; $TSS_{in(\geq x)}$ is the TSS in the influent greater than or equal to X µm (mg/L); and $TSS_{eff(\geq x)}$ is the TSS in the effluent greater than or equal to X µm (mg/L).

The calculated removal efficiencies are shown in Table 1 for particle size fractions above and below 200, 100, 75, 50, 25, and 10 µm for both pavement types. It is emphasized that these fractional removal efficiencies are valid only for the specific influent that was used in these simulated experiments. If an influent with a different PSD were applied to the pavements, the removal efficiencies for the various size fractions would vary accordingly.

Of particular importance, from the perspective of stormwater treatment in Calgary, are the particles over 75 µm. As shown in Table 1, the removal efficiency for particles greater than or equal to 75 µm is above 99% for both pavement types for this particular type of sub-250-µm street sweeping influent. For particles finer than 75 µm, the removal rates are 74% for the Eco-Stone and 83% for the porous asphalt. As the size range decreases from 75 µm, the removal rate begins to drop drastically for both pavement types. From a water quality standpoint, the performance of both pavement types in this study for removing solids is exceptional. Again, it must be stressed that these values are only valid for the influent used for this study. For future studies it would be very informative to apply influents with different PSDs and observe the changes in removal efficiencies for various influent size ranges.

TSS Removal Characteristics Observed in the Laboratory Experiments

The long-term effluent TSS concentrations for both pavements are shown in Fig. 8. Removal efficiencies for both surface types were 96% averaged over 100 data points, and with standard deviations of 2.8 and 3.6% for the Eco-Stone and porous asphalt models, respectively.

The particle size distributions of the influent and effluent for the laboratory Eco-Stone and porous asphalt models are shown in Fig. 9. The effluent particle size distribution results for the sub-250-µm street sweepings were very similar for both pavement types, and were also very similar for the laboratory and field experiments at Currie Barracks. The fractional removal efficiencies for the laboratory installations are shown in Table 2. The results are very similar to those observed in the field installations. Again, the removal efficiency decreased for the finer fractions.

For fractional removal efficiencies in the very fine range (i.e., <10 µm), the accuracy of this method becomes questionable. To illustrate the reason behind this, consider that, in the influent, there were consistently zero particles detected below 1 µm, whereas for the effluent, the average fraction below 1 µm was as high as 25%. It would appear as though fines, which were not present in the influent, are appearing in the effluent. The reason for this apparent slight discrepancy for small particle sizes is due to the precision of the PSD analyzer. The effluent was much finer than the influent, and was also only 5% of the influent concentration. Therefore, very fine elements that may have been obscured due to very low absolute quantities of particles in the influent, were observed in the effluent because of an "amplification effect" of the finer particles. The particles were present in the influent, but they were in such minute quantities that they were below the detectable limits of the analyzer, or otherwise obscured due to low particle numbers.

TABLE 1. Particle size fractional removal efficiencies for the Eco-Stone and porous asphalt installations at Currie Barracks[a,b]

Size fraction (µm)	*Proportion of influent under size fraction (%) n = 3*	*Standard deviation (%) n = 3*	*Proportion of effluent under size fraction (%) n = 9 (6)*	*Standard deviation (%) n = 9 (6)*	*% Removal for size fraction*
>200	30.5 (27.8)	8.2 (4.5)	4.5 (2.1)	2.6 (1.2)	99
>100	66.2 (64.6)	5.2 (3.8)	7.1 (3.6)	2.8 (2.4)	99
>75	75.4 (74.2)	2.5 (3.9)	9.0 (4.8)	3.1 (3.5)	99
>50	82.9 (81.7)	1.3 (4.2)	12.5 (7.1)	3.7 (5.1)	99
>25	89.8 (88.4)	0.9 (2.7)	20.7 (12.8)	4.8 (6.9)	98 (99)
>10	94.9 (94.5)	0.2 (0.8)	37.8 (29.2)	7.8 (6.6)	97 (97)
<200	69.5 (72.2)	8.2 (4.5)	95.4 (97.9)	2.6 (1.2)	90 (94)
<100	33.8 (35.4)	5.2 (3.8)	93.3 (96.4)	2.8 (2.4)	80 (87)
<75	24.6 (25.8)	2.5 (3.9)	91.0 (95.2)	3.1 (3.5)	74 (83)
<50	17.1 (18.3)	1.3 (4.2)	87.5 (92.9)	3.7 (5.1)	64 (76)
<25	10.3 (11.5)	0.9 (2.7)	79.3 (87.2)	4.8 (6.9)	45 (64)
<10	5.1 (5.5)	0.2 (0.8)	62.2 (70.9)	7.8 (6.6)	13 (40)

[a] Porous asphalt numbers are shown in parenthesis after the Eco-Stone numbers.

[b] Similar numbers are not repeated.

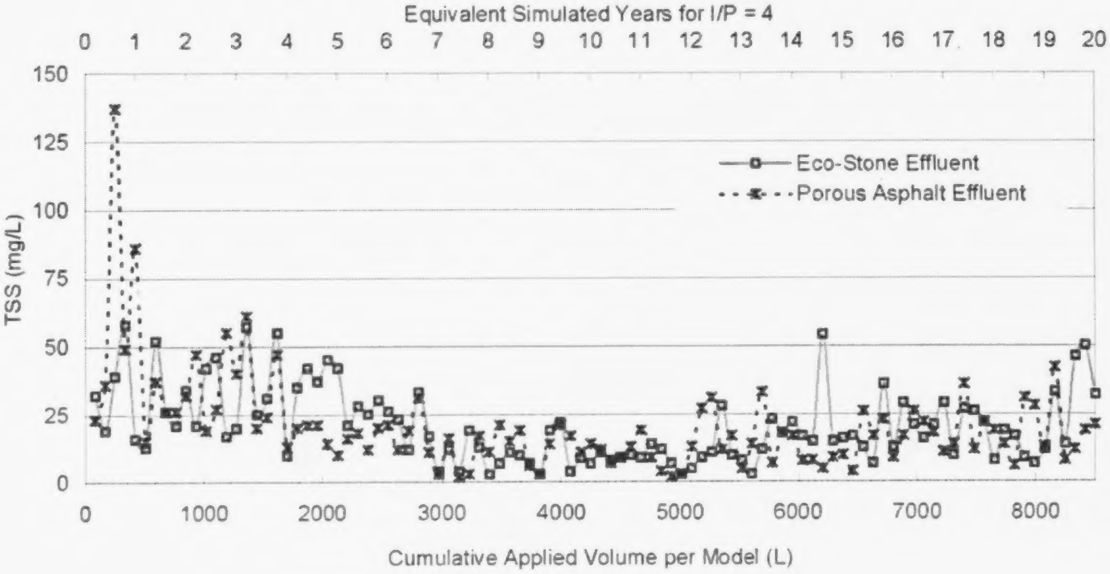

Fig. 8. Long-term effluent TSS for Eco-Stone and porous asphalt.

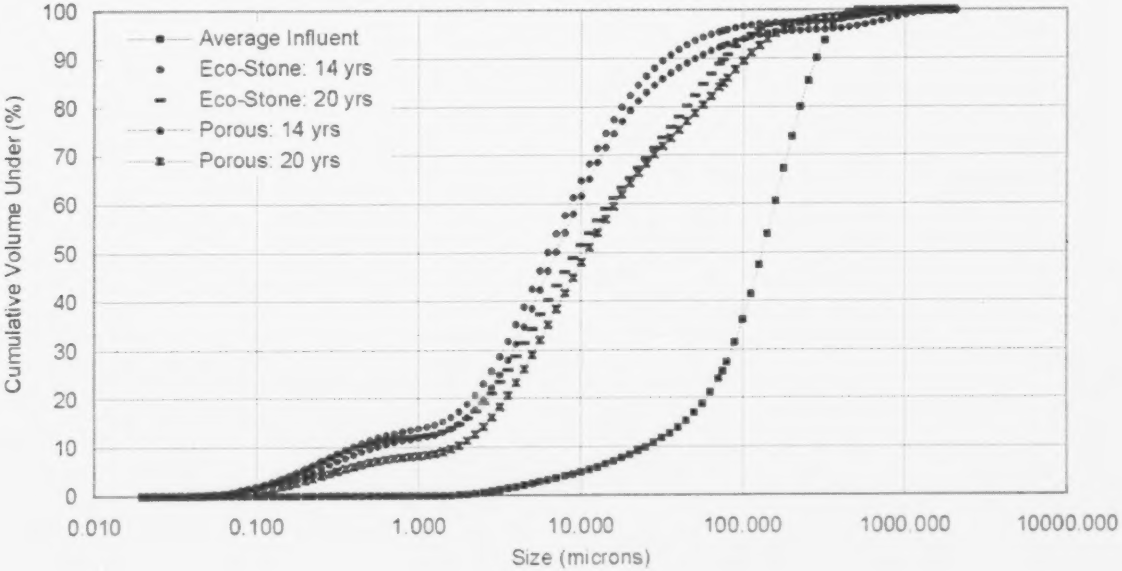

Fig. 9. Particle size distribution for the influent and effluent for long-term laboratory simulation for sub-250-μm street sweepings.

TABLE 2. Particle size fractional removal efficiencies for the Eco-Stone and porous asphalt models for the long-term laboratory simulations

Size fraction (µm)	*Proportion of influent under size fraction (%) n = 10*	*Standard deviation (%) n = 10*	*Proportion of effluent under size fraction (%) n = 9*	*Standard deviation (%) n = 9*	*% Removal for size fraction*
>200	24.7	3.9	3.8 (4.3)	2.7 (2.2)	99
>100	62.0	3.8	7.1	2.5	99
>75	72.9	3.3	7.7 (9.0)	3.6 (2.1)	99
>50	82.2	2.8	11.1 (12.3)	4.8 (4.2)	99
>25	89.8	1.4	20.1 (20.7)	7.2 (6.2)	99
>10	95.1	0.4	41.0 (42.3)	8.5 (9.0)	98
<200	75.3	3.9	96.2 (95.7)	2.7 (2.2)	95
<100	38.0	3.8	92.9	2.5	90
<75	27.1	3.3	92.3 (91.0)	3.6 (3.1)	86
<50	17.8	2.8	88.9 (87.7)	4.8 (4.2)	79 (80)
<25	10.2	1.4	79.9 (79.3)	7.2 (6.2)	68
<10	4.9	0.4	59.1 (57.7)	8.5 (9.0)	51 (52)

[a] Porous asphalt numbers are shown in parenthesis after the Eco-Stone numbers.

[b] Similar numbers are not repeated.

Size Distribution of Particle Accumulation Throughout the Structure

For the laboratory experiments, Fig. 5 and 6 show the results for the PSD analyses of the various layers of the pavement models, with the layer locations shown in the upper corner of the figures. For both pavement models, the layers above the geotextile appear to have very little influence on the sieving of specific particle sizes, while the geotextile itself appears to be most influential. The PSD of the sediment changes very little through the surface course and bedding course, although it is followed by a relatively drastic change above and below the geotextile (Locations D and E for Fig. 5, and Locations C and D for Fig. 6). The material is substantially finer below the geotextile as compared with the material above the geotextile. Interestingly, there also appears to be a change from the top to the bottom of the base course material. Regardless, these findings are significant in that they demonstrate the role the geotextile can have in filtration for permeable pavements.

At the time of this study, the literature contained very few studies that examined the particle sizes removed in permeable pavements and exactly where in the pavement structure those particles were being removed. The literature contains many studies discussing total efficiency of TSS removal, but these studies did not focus on the influent composition and the effluent PSD. Furthermore, there is very little information on the long-term performance of these pavement structures due to costs and wait periods associated with long-term studies. The lab experiments were an attempt to circumvent the long time periods, but to still gain a relatively good idea of long-term performance. Thus, the laboratory studies complemented the field studies; however, as with any laboratory study, the utility of the results depends on how well the real-world conditions leading to clogging and solids removal were simulated in the laboratory. The capacity for particles to pass through the upper layers of the pavement decreases slowly and progressively with time as finer and finer particles are trapped until a relatively impermeable matrix (or "crust") is formed (Balades et al. 1995; James and Gerrits. 2003; James 2004). This "clogged" area is usually limited to the upper few centimetres of the surface structure (Balades et al. 1995; Pratt et al. 1995; James and Gerrits 2003; James 2004) and is therefore characterized by an increase in the quantity of material retained in the upper surface, and not necessarily by migration of sediment particles to the subsurface layers of the structure (Balades et al. 1995). The laboratory experiments did not mimic the wetting and drying cycles and vehicular traffic impacts that create conditions leading to the formation of that crust; however, the overall removal performance was comparable. In addition, oil and grease effects were not incorporated in the laboratory model. Oil and grease have been shown to contribute to reductions in infiltration capacity with age (Gerrits 2001) and thus, possible differences between the laboratory and field results may be attributed to the lack of oil/grease in the laboratory models. The inclusion of these effects was beyond the scope of this study. While greater work is needed to improve the representativeness of the lab experiments, the approach described in this paper allows for further optimization of the composition of the pavement structure in future studies, especially when combined with optimizing the long-term hydraulic performance of these types of pavement structures.

Conclusions and Recommendations

The primary objectives of this research were to determine the performance of two types of permeable pavement with respect to hydraulics and water quality, and to try to gain a better understanding of the mechanisms and processes behind solids removal and clogging within permeable pavement structures. Results demonstrated that both permeable pavement types exhibited excellent TSS removal efficiencies both in laboratory and field settings, typically ranging from 90 to 96%. The effluent was found to be considerably finer than the influent, and the "sieving action" in a laboratory setting was found to occur most predominantly at the geotextile layer. This is somewhat contradictory to the findings of other studies (Balades et al. 1995; Pratt et al. 1995; James and Gerrits 2003; James 2004) in which filtration primarily took place at the surface of the pavement; this may be due to the influent characteristics and the lack of "crust" formation in the laboratory as it was not feasible to replicate the wetting and drying cycles and vehicular traffic impacts in the laboratory experiments. The findings in this study are significant in that they highlight the potential of being able to optimize the composition of the pavement structure by choosing appropriate materials and locations for the various courses and geotextiles in the pavement. Furthermore, 99% of particles sized 75 microns and over were removed by these particular pavement structures, whereas 74 to 86% of particles under 75 microns were removed. These fractional removal efficiencies are true only for the sub-250-μm sediment used in this study. Nonetheless, they show that the pavements meet current City of Calgary TSS removal guidelines.

Recommendations for future research include studies on the effects of influent TSS concentration on overall TSS removal rates and on fractional solids removal efficiencies of these pavement structures using a variety of influent sediments, with multiple particle size distributions.

Experiments examining the influences of different joint fill and bedding materials, as well as different geotextiles, and geotextile locations throughout the pavement structure, may provide more insight into what combination provides the best balance between solids removal and long-term hydraulic performance. Finally, realistic methods to simulate the wetting and drying cycles and the motion and impact of vehicular traffic are recommended as they would allow more representative long-term laboratory experiments to be performed.

Acknowledgments

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Effectiveness of Compost Biofilters in Removal of Sediments from Construction Site Runoff

Vahid Taleban,[1] Karen Finney,[1] Bahram Gharabaghi,[1]* Ed McBean,[1] Ramesh Rudra,[1] and Tim Van Seters[2]

[1] *School of Engineering, University of Guelph, Guelph, Ontario, N1G 2W1, Canada*
[2] *Toronto and Region Conservation Authority, Downsview, Ontario, M3N 1S4*

The effectiveness of compost biofilters in removal of suspended sediments from stormwater runoff was evaluated. Field experiments were conducted in the summer of 2006 at the Guelph Turf Grass Institute, University of Guelph, to verify the sediment removal efficiency of the compost biofilter from synthetic stormwater runoff. The average sediment removal efficiency of 8-inch (20-cm) compost biofilters (socks) for 5, 10, and 15 rolls were 34, 48, and 60%, respectively. The average sediment removal efficiency for 18-inch (45-cm) socks for 5, 10, and 15 rolls were 69, 84, and 95%, respectively. The decrease in sediment removal efficiency of the biofilter over time was significant. The average sediment removal efficiency of 5 rolls of the 18-inch (45-cm) sock started to decrease gradually from 70 to 62, 58, 56, and 54% after 1, 5, 10, 15, and 20 consecutive runs. Sediment removal efficiency of the biofilter for sediment particles in the size range of clay was found to be 30%, while for coarser particles such as fine silt and coarse silt was 50 and 80% removal efficiencies, respectively.

Key words: **compost, biofilter, stormwater, treatment, construction sites**

Introduction

Construction activities have significant potential to cause water pollution and stream degradation in areas under urban development if erosion and sediment control measures are not implemented properly. These activities could be considered a major source of sediment due to heavy land disturbance. Soil loss rates from construction sites can be 10 to 20 times that of agricultural lands (U.S. EPA 2000). The soil erosion rate from construction sites can range from minor amounts to 224 $t \cdot ha^{-1} \cdot yr^{-1}$ depending on climate, soil type, slope, and best management practices (NCDENR 2000). Runoff originating in construction sites may have a turbidity range of several hundreds to several thousands nepholometric turbidity units (NTU) despite use of proper management practices (Minton and Benedict 1999).

Compost from Canadian sources has not yet been tested for its effectiveness in stormwater runoff treatment since using compost as a biofilter for removal of suspended sediments, and sediment-bound contaminants from stormwater runoff is a relatively new idea. The objectives of this research on compost biofilters was to: 1) determine through-flow capacity of the biofilter for hydraulic design of the system; 2) determine the effectiveness of the biofilter in removal of suspended sediments from stormwater runoff near construction sites; and 3) determine the longevity of the biofilters.

Literature Review

In 2000, a workshop presented by the Great Lakes Science Advisory Board assessing the status of nonpoint source pollution control in the Great Lakes basin identified that construction sites are significant sources of sediments to urban streams (Clarifica Inc. 2004). If left unchecked, runoff pollution from urbanizing watersheds, especially from construction sites, will increase sediment loads to receiving watercourses. To develop a sustainable solution for this problem, industries, governments, and nongovernment organizations in Ontario are in the process of evaluating and updating design criteria for controlling sediment transport in urban areas under development (Bradford and Gharabaghi 2004; Gharabaghi et al. 2006a).

Fish are highly sensitive to sediment-laden waters. Excessive turbidity blocks sunlight penetration, reducing photosynthesis by algae and aquatic plants and thus food production for aquatic life (Henley et al. 2000). Suspended sediments provide surfaces upon which other contaminants such as heavy metals and chemicals can be adsorbed (Clark et al. 2003). Total suspended solids (TSS) concentrations are often used as an indicator of stream water quality (OMOE 2003).

Sediment Control Measures

The most common soil erosion and sediment control measures used on construction sites include rock check dams, silt fences, triangular silt dykes, baffles, straw mulching, sediment basins/traps, and skimmers. Factors such as the intensity and duration of storm events,

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* Corresponding author: bgharaba@uoguelph.ca

topography, and soil type can affect the sediment trapping efficiency of various control measures.

A study conducted by Schueler and Lugbill (1990) indicated that the TSS measured in runoff from construction sites were almost four times greater than the median value for varying storm conditions. They observed a removal efficiency of 46% of incoming sediment in outflows of sediment trapping devices, which was considered low due to the fine size of incoming sediment loads. Line and White (2001) monitored concentrations of TSS in outflow of three temporary sediment traps on two North Carolina construction sites, and observed sediment trapping efficiencies of 59 to 69% for sediment. Although these measures reduce the amount of pollutants entering the streams, they generally do not meet the required guidelines and standards (MOEE 1994).

According to Ward et al. (1979), trapping efficiencies greater than 90% are needed to meet typical water quality standards, but this efficiency is seldom attained. The European Inland Fisheries Advisory Commission (EIFAC 1965) reported that TSS concentrations above 80 mg/L are harmful to fish, and concentrations below 25 mg/L are tolerable. While several options exist to remove suspended solids from runoff, wet ponds are one of the common types of stormwater management systems in Ontario (OMOE 2003; Gharabaghi et al. 2006b). Settling is the primary mechanism for removal of TSS in construction sediment ponds, although physical and biochemical flocculation can be significant between rainfall events or during long residence times within ponds (OMOE 2003). According to the Ministry of the Environment's *Stormwater Management Planning and Design Manual*, treatment targets for post-construction stormwater ponds typically range from a minimum 60% removal to 80% removal of suspended solids (OMOE 2003).

A silt fence is a sediment-trapping practice utilizing a geotextile fabric (Tyler 2001) which is designed to increase the ponding depth (Goldman et al. 1986) to allow coarse sediment particles to settle out of storm runoff before it passes through the sediment barrier. Ponding depths increase as the silt fence geotextile becomes increasingly clogged by eroded sediment particles (Kouwen 1990). Barrett et al. (1998) concluded that effective sediment trapping by silt fence is not due to filtration by the fabric, but rather as a result of particles settling during detention behind the silt fence. Under low flow conditions, silt fences usually function well, but excessive runoff or ponding may lead to failure of the silt fence due to slumping and overtopping (Keener et al. 2006).

Compost Biofilters

Compost is commonly used for soil amendment, topdressing, and for erosion control, slope stabilization, and vegetative establishment applications (Faucette et al. 2005). Ettlin and Stewart (1993) found that yard waste compost can be used for slope stabilization and erosion control for slopes up to 42%. Yard waste compost was used in residential construction projects for erosion control and exhibited reduced sediment loads and improved water quality over conventional erosion and sediment control measures (Portland Metro 1994). Other compost related researchers (Michaud 1995; Demars and Long 1998; Glanville et al. 2001) reported that compost and mulch application has the capability to enhance the efficiency of existing erosion control management practices.

A new technology uses large volumes of compost material in mesh tubes also known as "socks" (diameters from 8 to 24 inches [20 to 61 cm]). These mesh tubes are very flexible multifilament polypropylene socks which are placed perpendicular to runoff for stormwater runoff treatment and sediment control (U.S. EPA 2006). In addition to ponding stormwater, these compost biofilters allow the water to flow through their three-dimensional matrix and filter it as water seeps through the organic media. The compost biofilter is designed to have a significant flow-through rate to reduce the chance of overtopping and water loss from under, or around, the compost biofilter.

After its lifetime, compost biofilter will be transported back to the composting facilities for recycling the contaminated sediments that were trapped in the biofilter. The sediments will be sieved out and properly disposed in a landfill. However, the compost media may be suitable for recycling through the composting facility.

The use of compost as a biofilter for treatment of stormwater runoff (as a through-flow medium to remove contaminants) is a relatively new idea. Ettlin and Stewart (1993) reported that compost filter berms improved sedimentation of TSS by 83% relative to bare soil, and by 72% relative to a silt fence over 5 natural rainfall events on a 34% slope. Additionally, the compost filter berm reduced TSS concentrations by 93% relative to bare soil, and by 91% relative to a silt fence. The U.S. Environmental Protection Agency (U.S. EPA 1997) tested a compost biofilter made of specially tailored leaf compost and reported a sediment removal efficiency of 90%, oil and grease removal efficiency of 85%, and heavy metals removal efficiency of 82 to 98%.

In another study at Ohio State University, flow-through rates of compost filter socks and a silt fence were evaluated, using a sediment-laden runoff concentration of 10,000 mg/L, containing only clay and silt, on a 20 degree slope for 30 minutes. The results showed that runoff flow-through rates of compost filter socks on average were 50% greater than the silt fence, and the ponding height behind a 24-inch (61-cm) silt fence was 75% greater than a 12-inch (30.5-cm) compost filter sock (Keener et al. 2006).

Faucette et al. (2006) conducted a series of laboratory tests on compost filter tubes and found that 12-inch (30.5-cm) and 18-inch (45-cm) compost filter socks have a TSS removal efficiency of 70% and a turbidity reduction of 84%, respectively.

Materials and Methods

Beginning in the spring of 2006, laboratory and field experiments were conducted at the University of Guelph, School of Engineering, and the Turf Grass Institute, to evaluate the effectiveness of the compost biofilters in removing suspended sediments from stormwater runoff. The compost was essentially made of various yard wastes including leaf, twigs, bark, and wood chips, and was provided from three major compost producers in southern Ontario: the Region of Peel, the Region of Waterloo, and Alltreat Farms. In this study we specifically used the coarser compost material by presieving the compost using a 0.5-inch (1.27-cm) sieve (also known as the "overs") to allow for higher flow-through capacity.

Physical Tests on Compost Material

Three representative samples per type of compost for a total of nine samples were tested for particle size distribution using an automatic shaker and a stack of varying numbered sieves, including sieves numbered 1 to 200. The void space was determined by placing 1,000 mL of compost into a graduated cylinder. Water was added to the cylinder to fill all the void spaces. The volume of water used to fill the spaces is equal to the volume of void spaces. Measurements were repeated three times and were done immediately after addition of water to the cylinder to prevent water being absorbed into the particles.

Flow-Through Tests

Flow-through tests were conducted to evaluate the flow-through capacity of the biofilter for the hydraulic design of the system. For this purpose, flow of the water passing through the filter sock under various hydraulic heads was measured. Figure 1 shows the flume used for flow rate tests. The flume was 1.5-m long by 0.69-m wide and 0.3-m deep with a constant head tank at the inlet end and a collection channel at the outlet. A prefilled and measured compost sock of 8-inch (20-cm) diameter was placed into the outlet of the flume and secured snugly along the bottom and sides of the flume. This was done to minimize the amount of water that exits without passing through the compost filter. Water was evenly distributed by using the water taps in the lab. Water depth was measured directly upstream and downstream of the compost sock. The upstream measurement was essential to obtain consistent flow rate. If the depth did not change for a period of five minutes, it was assumed that steady state had been reached and a flow measurement was taken.

To take a flow measurement, the pump was turned off and the siphon was broken manually by moving the pipe around. A stopwatch was used to determine the flow rate over a one minute period. The pump was then turned back on to drain the water from the bucket and the measurements were repeated two more times, for a total of three, at every constant depth.

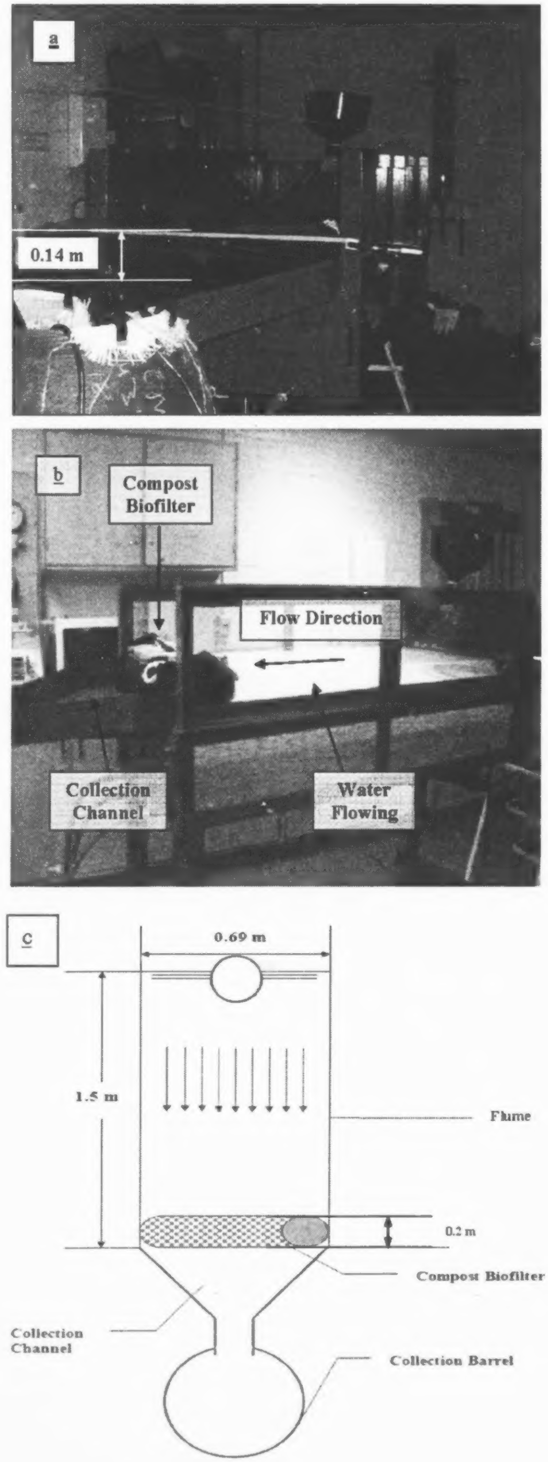

Fig. 1. a) Flume for flow rate testing, b) side view of the flume, and c) schematic drawing of the flume used for flow rate testing.

To achieve variable depths, the taps were turned down in stages to allow for readings at every 5- to 10-mm decrease in water depth. This test was performed three times for each compost type, with each run using a different sample, to account for the variability of the compost material.

Clean Water Tests

Compost filter tubes were subjected to clean water tests to determine if the biofilter would have an adverse effect on water quality due to wash-off of the compost material out of the sock. Clean water tests were performed in the same flume that was used for flow-through tests with the same setup. During the flow-through tests, samples were taken from the outflow water. The pH, TSS, turbidity, electrical conductivity, total Kjeldahl nitrogen, total phosphorous, and total organic carbon of the water filtered by the biofilter were all tested. The pH and conductivity measurements were done using digital readout probes. Turbidity was measured using a HACH 2100P Turbidimeter. TSS were measured according to the American Society for Testing and Materials standard method D3977-97C (ASTM 1999) using a 0.45-micron filter paper. Total Kjeldahl nitrogen, total phosphorous and total organic carbon were measured in the Soil & Nutrient Laboratory of University of Guelph using standard methods (APHA et al. 1992). Measured concentrations were compared with the Ontario Ministry of the Environment objectives (MOEE 1994) to evaluate their suitability for discharge to receiving waters.

Field Experiments

Field experiments were conducted in the summer of 2006 at the Guelph Turf Grass Institute, University of Guelph, to evaluate sediment removal efficiency of biofilters. Compost type, number of socks, sock diameter, and flow rate were tested to determine their effect on sediment removal efficiency. In the initial setup, two 10-m long, 1.2-m wide channels were constructed beside each other and were called plot A and plot B. Figures 2 and 3 present a picture and a schematic drawing of the plots. Initially, the sod layer was removed in the two plot sites and the ground was levelled. An end channel was constructed using sheet metal formed into a triangular spout with upright walls to direct the water. Sheet metal walls were then placed upright and perpendicular to each other to form the water column. The channels were then covered in plastic sheet wrap.

Water was supplied using a large constant head tank. In order to make synthetic stormwater runoff, a high clay content soil (approximately 50%, according to Fig. 2) was dried, grounded, and sieved. For each run a soil-slurry was prepared by mixing a 2-kg mass of sieved soil with 40 L of clean water in a mixing column. The prepared slurry was mixed with the clean water and was delivered at the inlet of the channel at a set rate using peristaltic pumps into a 1.2-m wide perforated polyvinyl chloride pipe where it was first diluted and then well mixed with the steady-rate inflow of clear water at the weir box (used to distribute flow evenly across the plot) upstream of the plots. The target influent TSS concentration was 850 to 900 mg/L, which is consistent with the monitoring data collected in collaboration with the Toronto and Region Conservation Authority from previous studies on construction sites in the Greater Toronto Area (Gharabaghi and Fata 2006).

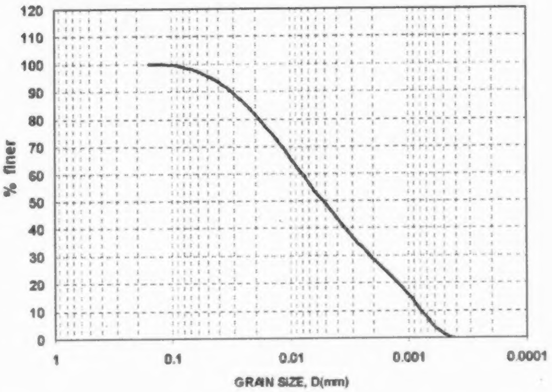

Fig. 2. Particle size distribution of the soil used for making synthetic stormwater.

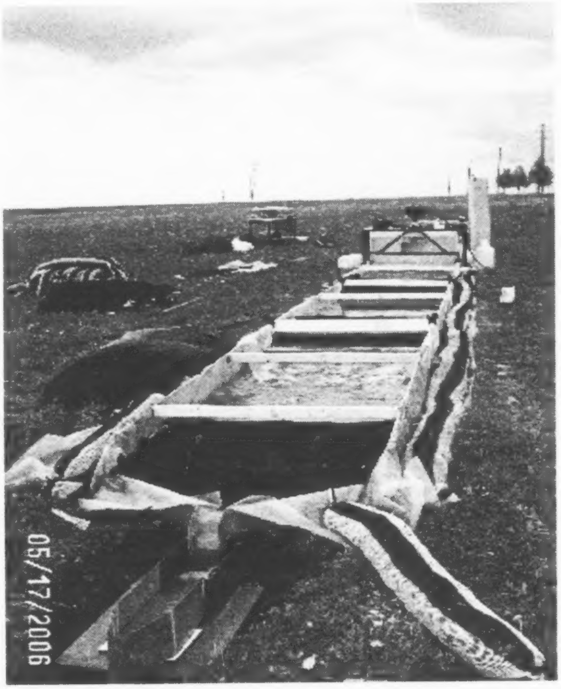

Fig. 3. The test plots at the Guelph Turf Grass Institute, including the three tier setup.

New filter tests were conducted in plot A. After the initial setup, three sets of compost biofilters were placed along the plot after constructing a three tier/level using plywood and gravel. Five 8-inch (20-cm) socks were placed on each tier for a total of fifteen socks (Fig. 3). Six runs were conducted on each sample of compost. Each run started with about 30 minutes of clear flow to establish steady state conditions followed by 30 minutes of turbid-water flow. Samples were collected towards the end of the run. Four samples were taken from each sampling location (I, Z, K, and O; Fig. 4) for a total of sixteen samples from each run. Then five 18-inch (45-cm) socks were placed at the bottom of the run, and flow rates of 0.5, 1.0, 1.5, and 2.0 L/s were tested with two runs per flow rate. Four samples were collected from each of the input and output locations.

Plot B was used to test the overall longevity of the system. Peel, Waterloo, and Alltreat compost biofilters were placed on the first, second, and third tiers respectively. Seventeen runs were performed and four samples were taken from each sampling location (Fig. 3) per run. The second longevity test performed on Plot B consisted of running 30 consecutive runs using five 18-inch (45-cm) socks filled with Alltreat compost. Flow had a constant value of 1 L/s during the longevity tests. Again, four samples were collected at each location. A summary of test data is presented in Table 1.

Runoff Sample Analysis

The TSS for runoff samples were analyzed in the fluids lab of the School of Engineering in the University of Guelph according to ASTM standard method D3977-97C (ASTM 1999). Three samples out of a total of four collected at each location were used in TSS measurement. Statistical analysis was conducted on sediment removal efficiencies by using PROC MIXED within Statistical Analysis Software (SAS) version 9, which fits a variety of mixed linear models to data and enables statistical inferences about the data. The response variable was outlet sediment removal efficiency. The four fixed-effect treatments were sock size, compost type, number of socks, and flow rate.

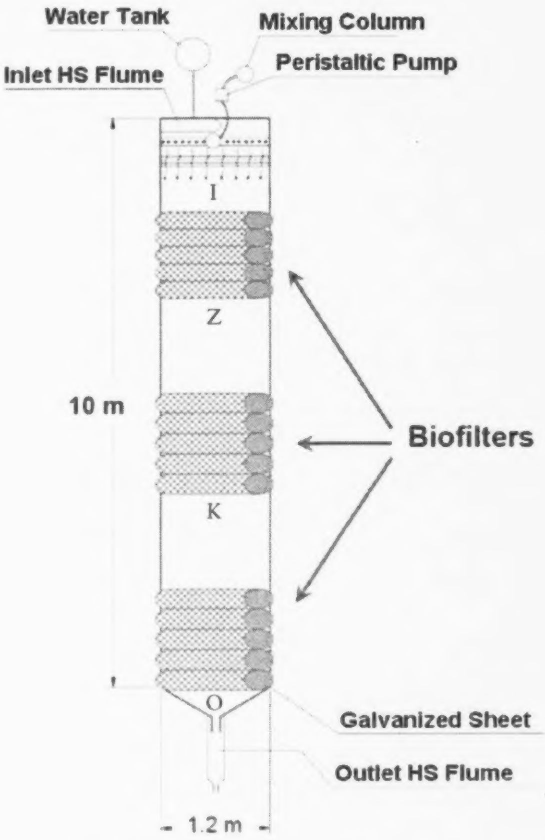

Fig. 4. Schematic drawing of plots including sample collection sites (I, Z, K, and O).

TABLE 1. Summary of field experiment data

Plot	*Sock size [inch (cm)]*	*Compost type*	*Average TSS concentration of influent (mg/L)*	*SD*[a]	*Average TSS Concentration of effluent (mg/L)*	*SD*
A	8 (20)	Waterloo	992	214	495	201
A	8 (20)	Alltreat	837	289	456	197
A	8 (20)	Peel	972	68	628	128
A	18 (45)	Waterloo	906	103	369	95
A	18 (45)	Alltreat	838	141	400	85
A	18 (45)	Peel	765	247	403	156
B	8 (20)	Hybrid	1,022	189	484	90
B	18 (45)	Alltreat	853	152	490	96

[a] SD = standard deviation.

Using a particle size analyzer (Mastersizer 2000), the fourth sample taken from each of the runs was run through the machine to verify any trends in the particle size distribution of each sample. The Mastersizer works by using the optical unit to capture the actual scattering pattern from a field of particles. It then calculates the size of the particles that create the pattern using the Fraunhofer model as well as the Mie theory.

Results and Discussion

Physical Tests

Figure 5 shows the average particle size distribution of the three compost types. Approximately 92 to 99% of compost particles passed through a 25.4-mm sieve, 82 to 94% passed through a 19-mm sieve, and 58 to 84% passed through a 9.42-mm sieve. The calculated uniformity and gradation coefficients both show that the composts are fairly well graded. For all three samples, void spaces ranged from 60 to 70%. All three compost types showed similar trends during the flow-through tests. Figure 6 displays the results of the flow rate tests after averaging for all replications. The average flow-through capacity per unit width of the 8-inch (20-cm) sock for the three compost materials without overtopping was 1.5 $L \cdot s^{-1} \cdot m^{-1}$. The flow-through capacity without overtopping of the 18-inch (45-cm) sock was approximately double the flow-through capacity of the 8-inch (20-cm) sock. Considering that the 8-inch (20-cm) sock when installed flattens to an oval shape with a height of about 14 cm, the seepage flow-through velocity was approximately 0.01 m/s.

Clean Water Tests

Clean water tests were conducted by measuring the TSS, pH, turbidity, electrical conductivity, total Kjeldahl nitrogen, total phosphorous, and total organic carbon. The suspended solid concentrations of the Alltreat and Waterloo composts were quite similar. Peel had slightly higher TSS concentration in the first 30 minutes. This is likely due to the release of fine particles.

Figure 7 displays these results clearly. To meet the target water quality guideline for the protection of aquatic life, TSS should have a threshold value of 25 mg/L for chronic exposure, and 80 mg/L for acute short-term exposure (EIFAC 1965). As shown in Fig. 7, a 10-min flush period will be required to meet these guidelines for short-term exposure.

Turbidity was also similar for all three compost types. After the first 10 minutes all turbidity approached zero (Fig. 8).

The pH of runoff met the Provincial Water Quality Objectives set by the Ontario Ministry of Environment and Energy (MOEE 1994), and was well within the 6.9 to 7.2 range. Results for total Kjeldahl nitrogen showed that concentrations approached zero after about 5 minutes

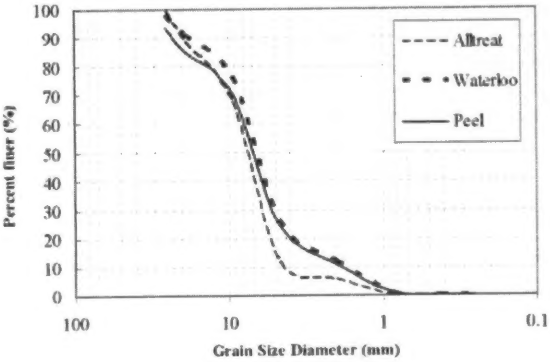

Fig. 5. Average particle size analysis of compost types.

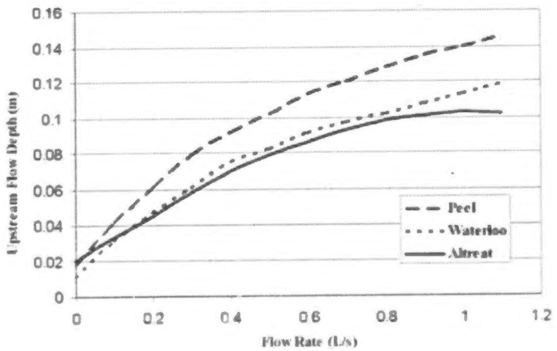

Fig. 6. Flow rate tests results after averaging for all replications.

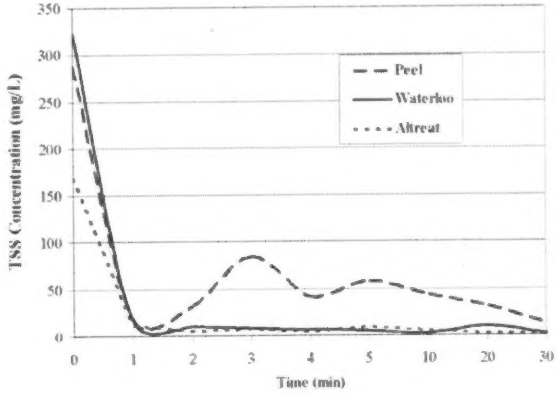

Fig. 7. Suspended solid concentration versus time comparison.

of clean water wash. Total phosphorous concentrations dropped below the detection limit (0.050 mg/L) after about 5 minutes of clean water flush through the biofilter. For total organic carbon, the three different composts followed the same trend, as shown in Fig. 9. After about 5 minutes of clean water wash, the concentrations ranged between 0 and 7 mg/L.

Field Experiments

Field experiments were divided into two categories. The first experiment examined the effect of sock size, number of socks, compost type, and flow rate on the removal efficiency of the biofilters. The second experiment investigated performance of the biofilter as void spaces became increasingly filled with sediment over time. A total number of 318 samples were taken during the first experiment, and 492 samples were taken during the second experiment. All samples were analyzed for TSS. The effect of number of socks and compost type on sediment removal efficiency for 8-inch (20-cm) socks is shown in Fig. 10.

Statistical Analysis

Using Statistical Analysis Software (SAS), data from experiment 1 were analyzed and a linear mixed model was chosen to study the effects of four factors on sediment removal efficiency. These factors included sock size (8 or 18 inches [20 or 45 cm]), compost type (Waterloo, Alltreat, or Peel), number of socks (5, 10, or 15) and flow rate (0.5, 1, or 2 L/s). The outlet sediment removal efficiency was calculated as the ratio of sediment concentration difference (inflow/influent – outflow/effluent) over the inflow/influent concentration.

The linear mixed model for experiment 1 includes the above four treatments, and their interaction as a fixed effect. The date and the run number are taken as random blocks to further reduce error. The residuals of the final model are normally distributed. The model was simplified by removing those nonestimable interactions and nonsignificant main effects. Details for calculations of the statistical analysis to determine the significance of each fixed effect on sediment removal efficiency are presented in Table 2. It was found that while sock size and number of socks had a significant effect on sediment removal efficiency, the compost type and flow rate did not. The flow rate is typically an important parameter, but over the narrow range of flow rates tested (0.5 to 2 L/s), the effect on removal efficiencies was not significant.

The model was also used to estimate the mean removal efficiency of different combinations of number of socks and sock sizes. The estimated mean removal efficiencies and 95% confidence intervals are presented in Table 3. For the 8-inch (20-cm) sock size, the mean removal efficiency increased linearly from 34 to 60% as the number of socks increased. The same condition applies to the 18-inch (45-cm) sock size; the removal

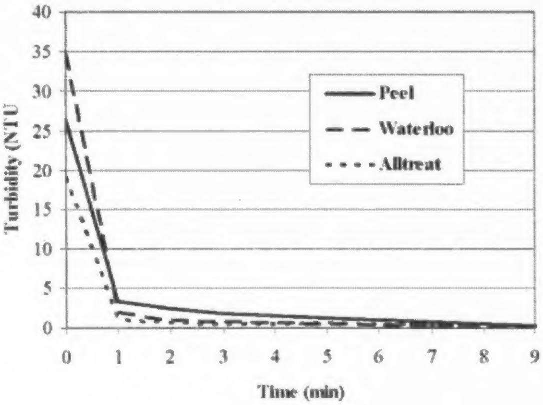

Fig. 8. Turbidity versus time comparison.

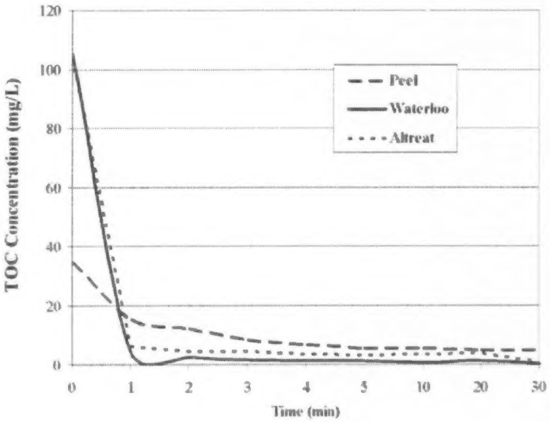

Fig. 9. Total organic carbon comparison.

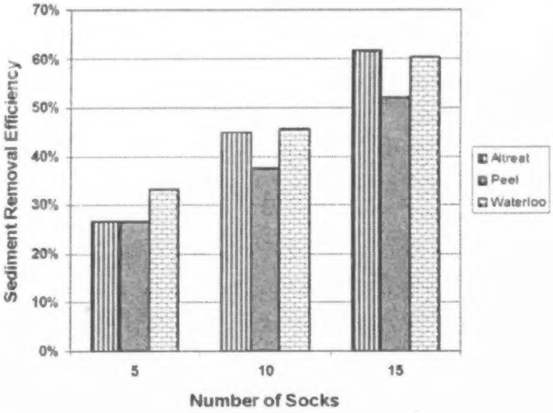

Fig. 10. Effect of number of socks on sediment removal efficiency.

TABLE 2. Type 3 tests of fixed effects to determine their significance on sediment removal efficiency

Effects	*DF*[a]	*F value*	*p value*
Sock size	1	4.56	0.0376
Compost type	2	0.01	0.9862
Number of socks	2	35.61	<0.0001
Flow rate	3	0.52	0.6703

[a] DF = degrees of freedom.

TABLE 3. Estimated mean values and 95% confidence intervals of sediment removal efficiency

Sock size [inch (cm)]	*Number of socks*	*Means*	*LCI*[a]	*UCI*[b]
18 (45)	5	0.69	0.39	0.99
18 (45)	10	0.84	0.54	1
18 (45)	15	0.95	0.64	1
8 (20)	5	0.34	0.12	0.55
8 (20)	10	0.48	0.27	0.69
8 (20)	15	0.60	0.38	0.81

[a] LCI = lower confidence interval.
[b] UCI = upper confidence interval.

efficiency increased linearly from 69 to 95% with the number of socks increasing from 5 to 15.

Experiment 2 was conducted to study the longevity of different types of socks and determine their approximate life expectancy. The treatments involved were the same as in experiment 1, except the flow rate was fixed at 1 L/s. Longevity test results for five socks of both 8 inches (20 cm) and 18 inches (45 cm) are presented in Fig. 11. The data from experiment 2 were analyzed by fitting separate time curves for different treatment combination groups. The response variable is the outlet sediment removal ratio. In this experiment, the treatment flow rate was fixed at the 1 L/s level, so we examined three fixed-effect treatments of sock size, compost type, and number of socks in our model. According to the analysis results, it was concluded that the time trend is best described by a quadratic curve, indicating that the removal efficiency decreases very fast for the first several runs, but gradually stabilizes at a certain level. Based on the model, the estimated mean and 95% confidence interval for 5, 10, 15, and 20 runs is presented in Table 4. There were only 17 runs for the 8-inch (20-cm) sock size group, which makes the prediction unreliable for the 15th and 20th runs; therefore, mean removal efficiencies for 8-inch (20-cm) socks were estimated for up to 10 runs. For 18-inch (45-cm) socks, having five socks, the expected removal efficiency decreased to approximately 25% after 20 runs, with range of 49 to 59% at the 95% confidence level.

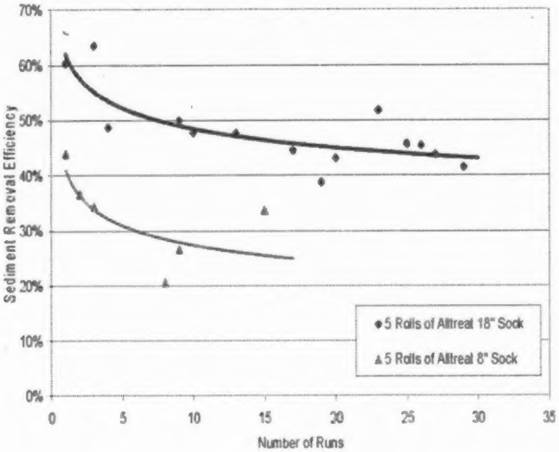

Fig. 11. Longevity test results in removing sediments.

TABLE 4. Estimated mean values and 95% confidence intervals of sediment removal efficiency

Sock size [inch (cm)]	*Run number*	*Means*	*LCI*[a]	*UCI*[b]
18 (45)	1	0.66	0.59	0.73
18 (45)	5	0.62	0.57	0.67
18 (45)	10	0.58	0.53	0.63
18 (45)	15	0.56	0.50	0.61
18 (45)	20	0.54	0.49	0.59
8 (20)	1	0.40	0.34	0.45
8 (20)	5	0.25	0.21	0.28
8 (20)	8	0.19	0.14	0.23
8 (20)	10	0.17	0.13	0.21

[a] LCI = lower confidence interval.
[b] UCI = upper confidence interval.

Particle Size Distribution Effect

The particle size distribution for inlet and outlet samples was measured with the Mastersizer instrument, and the results are presented in Fig. 12. Sediments were classified into four particle size classes: class 1 consisted of particles finer than 5.75 μm (clay size particles), class 2 consisted of particles between 5.75 and 20 μm (fine silt), class 3 consisted of particles between 20 and 60 μm (coarse silt), and class 4 consisted of particles larger than 60 μm (fine and medium sand). Removal efficiency at each point was calculated for all four classes. Table 5 presents the mean removal efficiency for different sediment particle size classes. Sediment removal efficiency was about 38% for clay size particles (class 1), about 44% for fine silt (class 2), 67% for medium silt (class 3), and 34% for coarse silt (class 4). According to the U.S. EPA (1993), other sediment control measures, particularly silt fencing, remove less than 20% of silt and clay from the stormwater, while Barrett et al. (1993) observed that 92% of TSS were clay and silt. Therefore, the capability of compost biofilter in removing 38 to 70% of clay and silt particles further highlights it's effectiveness in removal of fine particles from stormwater.

Conclusions

The following concluding remarks are based on the laboratory and field experiment results on compost biofilters:

- The seepage flow velocity through compost media was about 0.01 m/s. That is, the flow-through capacity (without overtopping) per unit cross-sectional area perpendicular to the flow direction for the three compost materials (overs) tested was approximately 10 $L \cdot s^{-1} \cdot m^{-2}$.
- The flow-through capacity of the 18-inch (45-cm) socks was approximately 200% higher than the flow-through capacity of the 8-inch (20-cm) sock. That is, the seepage flow velocity was roughly the same at 0.01 m/s.
- Sediment removal efficiency increased with the number of socks; the average sediment removal efficiency of the 8-inch (20-cm) socks for 5, 10, and 15 rolls were 34, 48, and 60%, respectively.
- Larger diameter socks provided larger filter media and were more effective than the smaller diameter socks. The average sediment removal efficiency of the 18-inch (45-cm) socks for 5, 10, and 15 rolls were 69, 84, and 95%, respectively.
- The sediment removal efficiency reduces significantly over time. The average sediment removal efficiency of 5 rolls of the 18-inch (45-cm) sock steadily and gradually reduced from 70 to 62, 58, 56, and 54% after 1, 5, 10, 15, and 20 consecutive runs.
- Sediment removal efficiency did not depend on flow rate as long as the stormwater runoff did not overtop the biofilter.
- During the field experiments, for a range of 765 to 1,022 mg/L influent TSS concentrations, the effluent TSS concentrations ranged from 369 to 628 mg/L.
- Particle size distribution is an important design factor for the biofilter. Sediment removal efficiency of clay size material was about 38% while for fine silt it was around 44%, and for medium and coarse silt the removal efficiency was around 67%.

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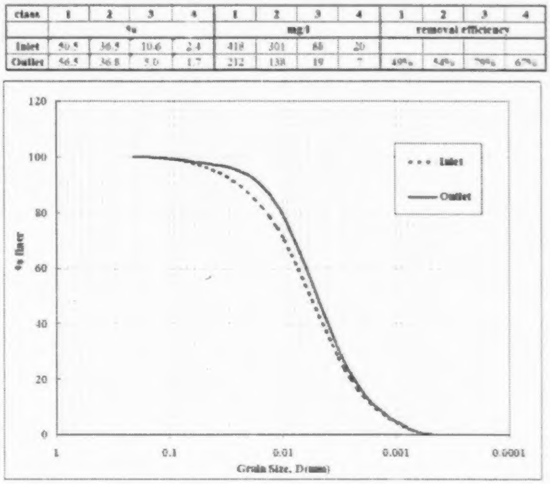

class	1	2	3	4	1	2	3	4	1	2	3	4
	%				mg/l				removal efficiency			
Inlet	50.5	36.5	10.6	2.4	418	301	88	20				
Outlet	56.5	36.8	5.0	1.7	212	138	19	7	49%	54%	78%	67%

Fig. 12. Particle size distribution of inlet and outlet samples. Class 1: 0.1 μm < d < 5.754 μm; Class 2: 5.754 μm < d < 19.953 μm; Class 3: 19.953 μm < d < 60.256 μm; Class 4: 60.256 μm < d < 1,096 μm.

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TABLE 5. Mean removal efficiency for different sediment particles size classes[a]

Class	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*	*1*	*2*	*3*	*4*
Sample	*%*				*mg/L*				*Removal Efficiency (%)*			
Mean Inlet	46.9	36.7	12.0	4.5	425	334	111	41				
Mean Outlet	51.8	36.3	6.9	5.1	266	187	36	27	38	44	67	34

[a] Class 1: 0.1 μm < d < 5.754 μm; Class 2: 5.754 μm < d < 19.953 μm; Class 3: 19.953 μm < d < 60.256 μm; Class 4: 60.256 μm < d < 1,096 μm.

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Sediment Assessment of Stormwater Retention Ponds within the Urban Environment of Calgary, Canada

Krista Westerbeek-Vopicka*

Strategic Planning and Policy, The City of Calgary, P.O. Box 2100, Station M, LOC #433 T2P 2M5, Calgary, AB, Canada

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The treatment of urban stormwater by retention ponds is known to be effective for water quality improvement as well as storm flow management and, in the past two decades, has become widely implemented. However, limited research has been conducted on the quality of sediment deposited in ponds. Therefore this study focuses on contaminant concentrations within the sediment from stormwater ponds built in Calgary, Canada. Electrical conductivity and the sodium adsorption ratio consistently exceeded the Canadian Council of Ministers of the Environment (CCME) agricultural soil quality guidelines, indicating a city-wide salt contamination issue. F3 hydrocarbon fractions, cadmium, chromium, copper, lead, selenium, and zinc were also identified as parameters of concern. In particular, the 61 Avenue SE duck pond displayed the greatest diversity and severity of contaminants due to the industrial catchment area. Removal and disposal options were limited due to the characteristics of the sediment. The examination of the solids content illustrated that all retention ponds will require the sediment to be dewatered prior to disposal. Disposal options were subsequently restricted to landfill disposal due to salt, metal, and/or hydrocarbon parameters exceeding CCME soil guidelines. One exception was the Deerfoot Trail and Highway 22X pond which could be directly disposed of in areas designated as commercial and industrial land use.

Key words: **disposal, heavy metals, hydrocarbons, retention pond, contaminants, sediment**

Introduction

Water quality is increasingly being researched due to growing water usage throughout Canada. Moreover, the distribution of Canada's water and population are creating pressures on water quality and quantity. Approximately 60% of Canada's useable freshwater drains towards the north, while 90% of Canada's population resides in the south (Environment Canada 1987). Although Canada generally has sufficient water to supply population demand, the distribution of people and fresh water creates localized pressures on water quantity and quality, especially in the lower populated portion of Canada. This is particularly important for urban centres as they have a significant impact on watersheds, predominately due to the effect of development increasing the amount of impervious groundcover which subsequently increases stormwater runoff (U.S. EPA 1993a; The City of Calgary 2000). Specifically, in southern Alberta, the Bow River sub basin managed under the South Saskatchewan River basin has a population density of 1 to 150 people/km^2. However, this then exceeds 1,000 people/km^2 within The City of Calgary limits (BRBC 2005). The dramatic increase in population density amplifies the percentage of impervious groundcover which in turn increases stormwater runoff. This consequently impacts the receiving watershed which is utilized by numerous downstream communities, reemphasizing the need for upstream management.

Urban runoff generally contains contaminants which include sediment, nutrients, metals, and hydrocarbons (U.S. EPA 1993a; The City of Calgary 2000). These contaminants are particularly important as they can cause severe impacts to a watershed. To address stormwater runoff, The City of Calgary has implemented the use of retention ponds for all new developments since they are known to be effective for water quality improvement and storm flow management (U.S. EPA 1993a; The City of Calgary 2000). Retention ponds have several mechanisms to remove contaminants. In particular, sediment is removed due to the deceleration of stormwater as it enters the retention pond, allowing the sediment to settle out of the water column by way of gravity (Kadlec and Knight 1996). Removal of sediment is important because it causes adverse affects to fish activity, and impairs spawning areas and benthic habitats (BRBC 2005). Nutrient removal focuses specifically on phosphorus and nitrogen since they drive eutrophication in receiving waters (U.S. EPA 1993a; Kadlec and Knight 1996). The uptake and release of nitrogen and phosphorus within a retention pond is dependent on water chemistry, sediment characteristics, and vegetation (Levine and Willard 1990). However, the primary method of removal from the water column is sedimentation with associated particulates (Kadlec and Knight 1996). Metals present in stormwater drainage systems typically include copper, chromium, cadmium, nickel, lead, iron, manganese, and zinc but may include other metals depending on activities present within the catchment area (U.S. EPA 1993a). Metals are important as they are persistent pollutants which can bioaccumulate in soil or plants and are most likely to influence sediment toxicity as they quickly concentrate (Mungur et al. 1995; Kadlec and Knight 1996; Heal 1999). Retention ponds are capable of removing metal pollutants through three

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* Corresponding author: krista.vopicka@calgary.ca

mechanisms. These include binding to soils, sediments, particulates, and soluble organics or precipitation as insoluble salts, or uptake by plants and bacteria (Kadlec and Knight 1996). Hydrocarbons are important due to the long half lives of the compounds and have been known to persist in water and soil matrices (Thurston 1999; Dabrowski et al. 2002). Runoff typically introduces these either as water-dissolved or particulate-associated when they enter the stormwater management system. Removal of hydrocarbons is achieved through several processes including volatilization, photochemical oxidation, sedimentation, sorption, and biological degradation (including fermentation, aerobic and anaerobic respiration) (Kadlec and Knight 1996). However due to the highly hydrophobic nature of hydrocarbons, sedimentation with associated particulate matter dominates as the main factor which removes hydrocarbons from the water column (Thurston 1999; Dabrowski et al. 2002; Braskerud and Haarstad 2003).

These internal removal processes illustrate two important points. Firstly, the primary removal mechanism of contaminants is sedimentation with associated particulates. Secondly, retention ponds continually accumulate sediment. It is also known that excessive sediment accumulation within retention ponds can lead to a reduction in water quality. This is due to the loss in storage volume, decreased velocity attenuation, and decreased sedimentation capabilities (Revitt et al. 1999). Therefore, to properly maintain treatment efficiencies of retention ponds, regular dredging is required (GIC Inc. 1999). Extensive research has been directed to the removal efficiency of retention ponds and the subsequent water quality leaving the stormwater system. However, limited research has been directed towards the sediment, the fate of the contaminants sequestered by the system, or the concentration of these contaminants within the retention ponds.

With the introduction of retention ponds being utilized as treatment systems within the stormwater network, applicable legislation was examined. Alberta Environment has adopted the federally issued Canadian Council of Ministers of the Environment (CCME) guidelines (CCME 1999) to supplement provincial legislation. Sediment within constructed wetlands cannot be regulated by criteria intended for natural wetland sediment. CCME sediment guidelines are intended to maintain biodiversity and organism abundance. Retention ponds are not designed to be pristine systems, replace removed wetlands, or be maintained as ecosystems. Instead, they are intended for the removal of sediment and to improve water quality. The accumulation of sediment however must be periodically dredged to maintain pond efficiency (Kadlec and Knight 1996; The City of Calgary 2000), and be disposed of offsite. With the required ex situ disposal of sediment and the lack of applicable sediment-specific criteria, any sediment removed from the retention ponds will hereafter be classified as a soil and subsequently regulated under CCME soil guidelines.

The primary objective of this paper was to provide a base of knowledge on the contaminant concentrations within the sediment in a variety of ponds receiving stormwater from differing residential, commercial, industrial, and highway land uses. Secondly, sediment removal and disposal options were evaluated.

Selection of the Study Sites

Stormwater retention ponds have been utilized by The City of Calgary since 1979 when Calgary constructed its first wet pond at 68 Street and 17 Avenue SE to moderate stormwater event volumes reaching the river systems. By 1988, retention ponds were designed as part of the storm drainage systems with a storage capacity to accommodate a 1-in-100-year rainfall event (The City of Calgary 2006a). During this time it was also recognized that retention ponds demonstrate beneficial water quality improvements as they are efficient at pollutant removal, capable of addressing multiple contaminants, are sustainable, require relatively low maintenance, have a high aesthetic appeal, and are cost effective (Mitsch and Gosselink 1993; Kadlec and Knight 1996; Magmedov et al. 1996; Griffin and Upton 1999; Lin et al. 2002). This has led to all new residential subdivisions requiring the installation of retention ponds to treat the stormwater prior to its discharge into the rivers.

Site Selection Criteria

The retention ponds chosen as sampling sites for this study were selected from Calgary wet ponds and wetlands which have been receiving stormwater for an extensive period of time and subsequently have developed a distinct sediment layer. The intention was to encompass retention ponds that were reaching their water treatment capabilities and would require dredging shortly to maintain treatment efficiency. Furthermore, sites were chosen to represent a variety of dominant land uses present within the catchment area, and represent degrees of ongoing development within Calgary. Based on these criteria, the following five stormwater facilities were selected:

Deerfoot Trail and Highway 22X pond (Reid Crowther & Partners Ltd. 2000) has been in operation since 2001 to treat runoff from a 64 ha area collecting solely highway runoff. Pond design specifics consist of a surface area of 7,950 m^2, a permanent water level (PWL) of 1,031.5 m, and a pond bottom elevation of 1,029.0 m.

68 Street SE retention lake became operational in 1979 and accommodates runoff from an exceptionally large catchment area of approximately 2,925 ha (The City of Calgary 2006b). The land use and approximate percentage present within the catchment area are residential (70%), parkland (10%), commercial (15%), and light industrial (5%). Pond design specifics available include the pond surface area of 20.0 ha, a pond bottom elevation of 1,047.6 m, and a PWL of 1,049.9 m.

Edgemont wetland has been in operation since 1996 (IMC Consulting Group Inc. 1995), servicing a designated drainage area of 114.2 ha. Land use consists of approximately 90% residential and 10% parkland. Edgemont wetland was designed with a PWL elevation of 1,175.5 m, a pond bottom elevation of 1,173.5 m, and total surface area of 17,768 m^2.

Harvest Lake construction was completed in 1988 and has been in operation for 20 years. It was designed to treat a drainage area of 390 ha (The City of Calgary 2006b). Land use consists of approximately 90% residential and 10% parkland. Pond specifics include a surface area of 44,587 m^2, a PWL of 1,066.0 m, and a set pond bottom elevation of 1,062.7 m.

61 Avenue SE duck pond became operational in 1985 and was designed to treat stormwater originating from a 28 ha catchment area consisting of solely industrial land use (Westhoff Engineering Resources Inc. 1998). However, due to growth of Calgary, the pond currently receives runoff from 1,375 ha. Pond design specifics encompass a surface area of 5,477 m^2, a pond bottom elevation of 1,032.0 m, and a PWL of 1,034.0 m.

Material and Methods

Sampling Events

All samples were collected from February 24 to 26, 2004 during the winter freeze. Sediment samples were collected in midwinter, which allowed the samples to be more accurately obtained and mapped. Additionally, winter collection avoided disturbance of the sediments that might occur if ponds were accessed by boat (paddle or poling action) and avoided drifting that would occur when utilizing a boat. Discrete grab samples were collected at even intervals from the inlet to the outlet of the pond, following the main flow path which was determined from the facility design plans. Grab samples were collected using an Eckman dredge, and appropriately preserved and kept on ice.

Methods

All of the parameters were analyzed using standard methods. Total solids, fixed solids, and volatile solids were measured using Method 2540G from *Standard Methods* (APHA et al. 1998). Particle size distribution was determined using the Mastersizer 2000 (Malvern Instruments 1998) which was equipped with a Hydro G sample dispersion unit. Heavy metal analysis was conducted by MAXXAM Analytics laboratory in Calgary, Alta. The metals were digested using the EPA Method 3050B (U.S. EPA 1996a), and the samples were subsequently analyzed for the elements of interest using the appropriate inductively coupled plasma atomic emission spectroscopy (ICP-AES) and inductively coupled plasma mass spectrometry (ICP-MS) instrumentation. Total Kjeldahl nitrogen (TKN) was analyzed by MAXXAM Analytics laboratory in Calgary, Alta. TKN was digested following the EPA Method 351.2 (U.S. EPA 1993b) and subsequently analyzed using a spectrophotometer. Total phosphorus (TP) was analyzed by MAXXAM Analytics laboratory following EPA Method 351.2 (U.S. EPA 1993b). Collection, preservation, and storage of samples were the same as that for TKN. Samples were analyzed for F2 and F3 hydrocarbon fractions following procedures outlined in the *Canada-Wide Standard for Petroleum Hydrocarbons in Soil – Tier 1* document (CCME 2001). Samples were cleaned in situ with silica gel and analyzed on the gas chromatograph using a DB1 column. Electrical conductivity, pH, and chloride were analyzed for all samples using field probes. Determination of free liquids for all samples was completed by obtaining a composite sample of all the sample points from each retention pond and completing the EPA Method 9095A (U.S. EPA 1996b) paint filter liquids test.

Results

Solids

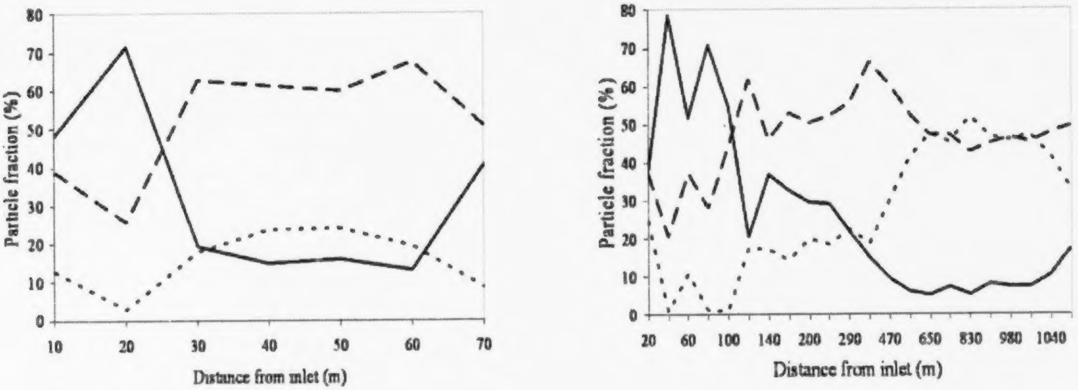
The total solids concentration by mass of each pond was found to have an average of 44.10, 30.34, 44.40, 41.83, and 38.48% for the Deerfoot Trail and Highway 22X pond, 68 Street SE retention lake, Edgemont wetland, Harvest Lake, and 61 Avenue SE duck pond, respectively. The highest average volatile solids concentration by mass was determined to be 4.5% for the 61 Avenue SE duck pond.

Particle Size Distribution

Deerfoot Trail and Highway 22X pond, 68 Street SE retention lake, Harvest Lake, and 61 Avenue SE duck pond (Fig. 1 A, B, D, E) all displayed a high proportion of sand at the inlet, while the Edgemont wetland displayed a consistent distribution of particle fractions throughout (Fig. 1C).

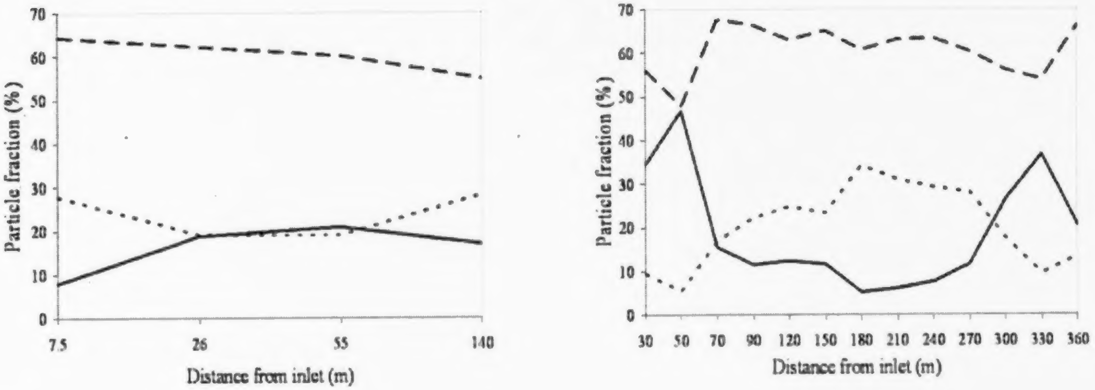
Metals

The main summary of the analyzed chemical constituents with their respective guidelines are displayed in Table 1. Chromium, copper, and lead mean concentrations exceeded CCME guidelines in the 61 Avenue SE duck pond, while cadmium and zinc mean concentrations exceed guidelines in both the 61 Avenue SE duck pond and the 68 Street SE retention lake. Furthermore, selenium concentrations exceeded guidelines in Harvest Lake and the 68 Street SE retention lake. An analysis of variance test and Tukey's test were utilized to identify differences



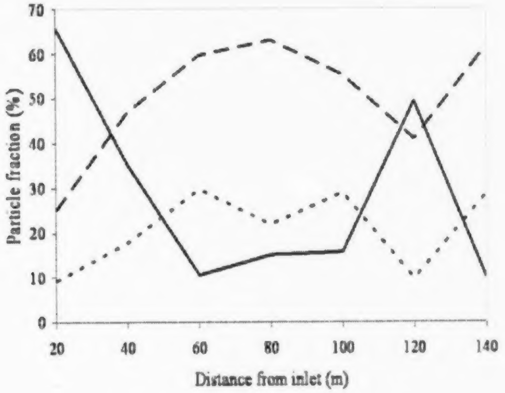
A: Deerfoot Trail & 22X pond

B: 68 St SE retention pond



C: Edgemont wetland

D: Harvest lake



E: 61 Ave SE Duck pond

Fig. 1. Particle fraction (%) of each fraction for sand (>50 microns), silt (50 to 2 microns) and clay (<2 microns). sand (—); silt (- -); clay (...)

TABLE 1. Total chemical concentrations for compounds with soil quality guidelines per sampling location[a,b]

		Concentration (mg/kg)										
		As	*Ba*	*Cd*	*Cr*	*Cu*	*Pb*	*Ni*	*Se*	*Tl*	*V*	*Zn*
CCME Guidelines	Agricultural	12	750	1.4	64	63	70	50	1	1	130	200
	Industrial	12	2,000	22	87	91	600	50	3.9	1	130	360
Deerfoot Trail & Hwy 22X	In	4	226	0.5	13	12	10	14	0	0.1	17	58
	Mid	6	250	0.8	21	22	14	31	0	0.3	29	118
	Out	4	204	0.6	15	15	10	18	0	0.2	22	80
68 St SE retention lake	In	6	231	1.4	38	48	78*	27	2*	0.2	26	347*
	Mid	6	275	1.5*	25	44	60	27	3*	0.2	29	307*
	Out	8	279	1.3	28	37	47	26	3*	0.2	32	240*
Edgemont wetland	In	7	330	0.9	26	28	17	32	1	0.2	32	151
	Mid	7	318	0.8	25	27	19	31	1	0.2	30	150
	Out	7	297	1.7	31	33	25	33	0	0.2	30	233*
Harvest Lake	In	4	339	1	20	18	12	24	3*	0.2	22	102
	Mid	4	282	0.8	23	27	18	29	11*	0.2	22	155
	Out	9	312	0.6	20	22	16	23	4*	0.2	26	114
61 Ave SE duck pond	In	4	195	20*	72*	44	78*	20	1	0.2	20	504*
	Mid	6	360	64*	192*	74*	106*	36	1	0.3	34	1,220*
	Out	6	375	64*	352*	89*	104*	38	1	0.7	34	1,110*

[a] As, arsenic; Ba, barium; Cd, cadmium; Cr, chromium; Cu, copper; Pb, lead; Ni, nickel; Se, selenium; Tl, thallium; V, vanadium; Zn, zinc.

[b] Bolded values with an asterisk (*) indicate mean values that exceed fine-grained CCME agricultural and industrial soil criteria (CCME 1999).

between ponds. Cadmium ($F_{0.05,4,13}$ = 14.11), chromium ($F_{0.05,4,13}$ = 6.65), copper ($F_{0.05,4,13}$ = 13.17), lead ($F_{0.05,4,13}$ = 45.82), and zinc ($F_{0.05,4,13}$ = 15.67) concentrations were determined to be significantly higher in the 61 Avenue SE duck pond compared with the remainder of the sites. Additionally, cadmium ($F_{0.05,4,13}$ = 14.11), zinc ($F_{0.05,4,13}$ = 15.67), and selenium ($F_{0.05,4,13}$ = 5.12) concentrations were determined to be significantly higher in the 68 Street SE retention lake, while only selenium concentrations were significantly greater in Harvest Lake ($F_{0.05,4,13}$ = 5.12) when compared with the remainder of the sites.

Nutrients

With respect to nutrients, nitrogen and phosphorus were examined. Specifically, TKN and TP were measured (Table 2).

TABLE 2. Total Kjeldahl nitrogen and total phosphorus mean concentrations of dredge samples per sampling location

Sample	*TKN (mg/kg)*	*TP (mg/kg)*
Deerfoot Trail & Hwy 22X	1,680.14	1,134.60
68 Street SE retention lake	3,332.27	1,185.81
Edgemont wetland	1,717.50	1,346.24
Harvest Lake	1,756.62	739.84
61 Ave SE duck pond	1,661.43	1,648.63

Hydrocarbons

Hydrocarbon analysis of the F2 fraction (Fig. 2) displays no significant difference between ponds ($F_{0.05,4,10}$ = 1.60). For all sites, the average concentration and ranges are well below the CCME agricultural soil F2 hydrocarbon fraction guideline of 900 mg/kg (Fig. 2).

With respect to the F3 fraction (Fig. 3), the hydrocarbon content of the 68 Street SE retention lake and the 61 Avenue SE duck pond were significantly higher than the remaining ponds ($F_{0.05,4,10}$ = 11.64). Figure 3 also illustrates that both the 68 Street SE retention lake and the 61 Avenue SE duck pond exceeded the allowable CCME agricultural soil F3 hydrocarbon fraction guideline of 800 mg/kg. Additionally, none of the sample points in any pond exceeded the industrial soil guideline of 2,500 mg/kg. However, it is important to note that the 68 Street SE retention lake and the 61 Avenue SE duck pond do approach the guideline.

Additional Sediment Parameters

With respect to the average pH values, all retention ponds were within the CCME soil quality guideline range of 6 to 8. The average pH of each pond was found to be 7.9 for the Deerfoot Trail and Highway 22X pond, 7.5 for the 68 Street SE retention lake, the Edgemont wetland, and Harvest Lake, and 7.4 for the 61 Avenue SE duck pond.

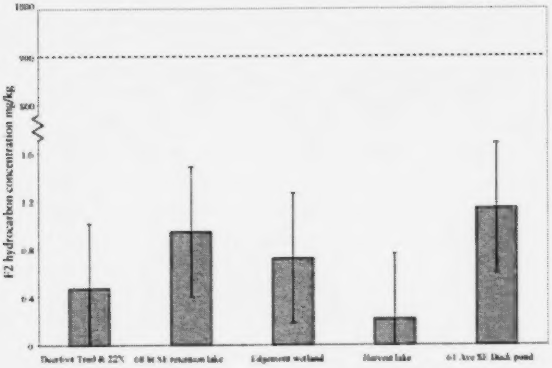


Fig. 2. F2 CCME hydrocarbon fraction concentrations of dredge samples per sampling site including CCME agricultural soil (dashed line) quality guidelines.

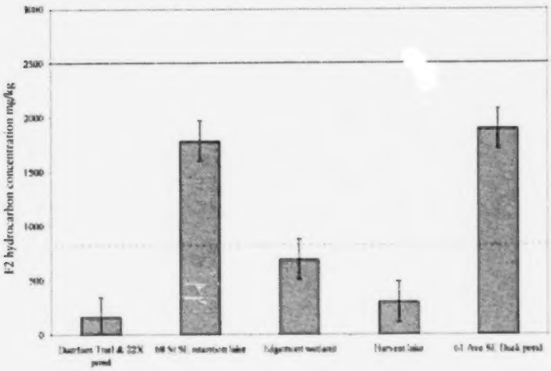


Fig. 3. F3 CCME hydrocarbon fraction concentrations of dredge samples per sampling site including CCME agricultural soil (dashed line) and industrial soil (solid line) quality guidelines.

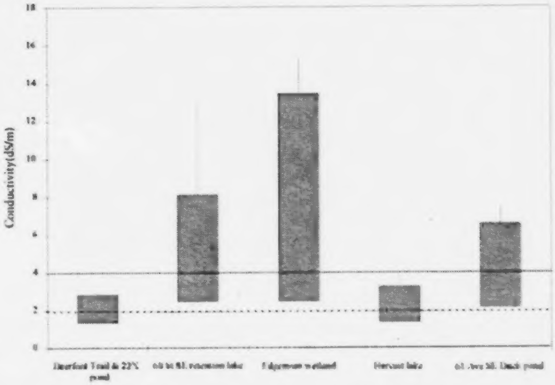


Fig. 4. Conductivity of dredge samples including CCME agricultural soil (dashed line) and industrial soil (solid line) quality guidelines.

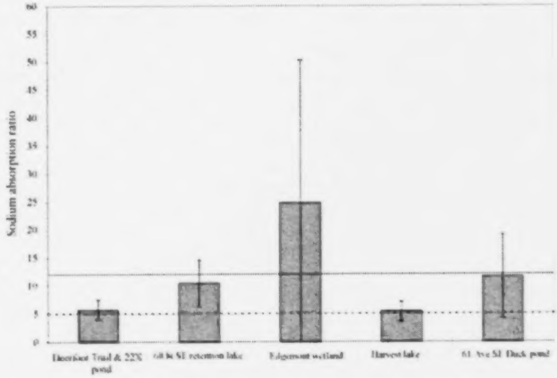


Fig. 5. Sodium absorption ratio of dredge samples including CCME agricultural soil (dashed line) and industrial soil (solid line) quality guidelines.

The electrical conductivity (EC) varied significantly between sites ($F_{0.05,4,48} = 6.11$), with measurements in the Edgemont wetland and 68 Street SE retention lake being significantly greater compared with those from the remaining sites. Additionally, all of the site averages exceeded agricultural soil guidelines, but only Edgemont wetland and 68 Street SE retention lake exceeded the industrial soil guideline (Fig. 4).

The average sodium adsorption ratio (SAR) of every retention pond exceeded the soil quality guideline of 5.0 for agricultural soil (Fig. 5). However, only Edgemont wetland exceeded the CCME industrial soil criteria of 12.0 (Fig. 5).

Paint Filter Test

Landfill criteria, regulated by Alberta Environment (1995), requires that any material disposed of within a landfill cannot have any free liquids present. To determine the presence of free liquids, a paint filter test was conducted utilizing EPA Method 9095A (U.S. EPA 1996b). Subsequently, results indicated that all sites contained free liquids.

Discussion

The primary objective was to determine the presence and concentration of contaminants in order to understand the extent of contamination of accumulated sediment within the retention ponds. Consequently, each parameter was evaluated against the CCME agricultural soil guideline, which was used as the baseline for comparison. This provided the most stringent guidelines and therefore is a suitable guideline to define between contaminated and noncontaminated soils. Additionally, parameter concentrations were evaluated against literature.

The parameters that exceeded the CCME soil guidelines in all ponds were EC and the SAR. This indicates that all of the retention ponds within The City of Calgary have salinity and sodicity contamination regardless of the catchment area differences. The major ions identified as contributors to the high EC were found to be chloride, sodium, calcium, and magnesium. The SAR also exceeded guidelines due to the larger proportion of sodium compared with calcium and magnesium. Examination of background soils present within the catchment areas illustrated limited areas of high saline soils, limited to 2.5% of the surrounding Calgary area (Kwiatkowski et al. 1996). This suggests that the increases in EC and the SAR are due to anthropogenic sources. This is important as The City of Calgary uses mainly sodium chloride, and to a lesser extent calcium chloride, for deicing roads (The City of Calgary 2006c), which potentially is the source for this widespread contamination. However, the presence of solonetzic soils within the Calgary area and the potential for natural salt leaching from the catchment areas cannot be discounted as a contributor to the high EC and SAR results (Wyatt et al. 1960; Macmillan 1987). The consistently exceeded guidelines also illustrates that salt is the primary contaminant of concern for this study area.

The remaining parameters varied between retention ponds, illustrating localized influences due to specific catchment area contribution differences. It was observed that of all the metals evaluated within this study, only cadmium, chromium, copper, lead, selenium, and zinc accumulated sufficiently to exceed the CCME agriculture guidelines in one or more ponds. Mean cadmium and zinc concentrations exceeded the CCME agricultural soil quality guidelines by 1.1 to 35 times for cadmium and 1.4 to 4.7 times for zinc in the 68 Street SE retention lake and the 61 Avenue SE duck pond, respectively. Chromium, copper, and lead exceeded guidelines by 3.2, 1.1, and 1.4 times, respectively, for the 61 Avenue SE duck pond. Additionally, selenium exceeded guidelines 3 and 6 times for the 68 Street SE retention lake and Harvest Lake, respectively. Literature reports of background soil levels (mg/kg) within Alberta are known to be 0.16, 0.01, 20, 12, and 74 for cadmium, chromium, copper, lead, and zinc, respectively (Knight and Klassen 2005). This indicates that anthropogenic sources are contributing to the contaminant loading of the sediment. Cadmium, chromium, copper, lead, and zinc concentrations were subsequently compared with other studies including Yousef et al. (1994a, 1994b); Kadlec and Knight (1996); Marsalek and Marsalek (1997); Heal (1999); Marsalek et al. (1999); Mallin et al. (2002); Vymazal and Krasa (2003); and Kamalakkannan et al. (2004). The metal concentrations in sediments from other urban retention pond studies have found concentrations (mg/kg) to range between 0.051 to 53.0 (cadmium), 0.97 to 128.0 (chromium), 0.45 to 1,441 (copper), 1.5 to 1,047 (lead), and 1.0 to 779 (zinc). These studies encompass diverse urban environments including residential, parkland, commercial, highway, and light industrial uses, similar to the diversity in urban land uses examined within this study. These concentrations illustrate two important points. Firstly, all of the retention ponds within this study had comparable concentrations to those observed in other urban retention pond sediments, with the exception of the 61 Avenue SE duck pond. Secondly, the ranges found in the literature emphasize the severity of the contamination observed within the 61 Avenue SE duck pond. This is especially true for cadmium, chromium, and zinc with respective average concentrations (mg/kg) of 49, 205, and 945, illustrating that sediment from this catchment area, dominated by industrial use, is severely contaminated compared with urban retention ponds within other studies.

TKN and TP concentrations are not regulated by the CCME soil quality guidelines. Literature reports levels of nitrogen-n in soil from Calgary's urban perimeter to be an average of 3,859 mg/kg for an A horizon, and 1,245 mg/kg for a B horizon (Macmillan 1987). Consequently, all nitrogen concentrations within the retention ponds are similar to the natural soil levels since average TKN concentrations within the ponds were found to range between these natural concentrations. Other studies have observed nitrogen concentrations between 108 to 464 mg/kg, and phosphorus concentrations between 9.6 to 164 mg/kg (Yousef et al. 1994a; Lazaridou-Dimitriadou et al. 2004) which are lower than 1,661 to 3,332 mg/kg and 740 to 1,649 mg/kg for nitrogen and phosphorus concentrations observed within this study. This illustrates that the nutrient contribution retained within the retention pond sediment is low within this and other studies. This provides minimal benefit when considering disposal options.

Hydrocarbons were anticipated to be present in urban areas due to fuel storage and automotive wear and maintenance (Allcock et al. 1991). The two most prominent hydrocarbon fractions found were the F2 (C_{10} to C_{16}) and F3 (C_{16} to C_{34}) fractions. F3 hydrocarbon anthropogenic sources are related to the handling, transport, storage, and disposal of heavy end fuel, oil, and grease products within urbanized areas (Allcock et al. 1991; Heal 1999). Meanwhile, F2 hydrocarbon can originate from the subsequent handling, transport, storage, and disposal of lighter end fuels. It was found that the F2 fraction was consistently well below the CCME agricultural soil guidelines, while F3 hydrocarbons were present in concentrations which exceeded guidelines for the 68 Street SE retention lake and 61 Avenue SE duck pond. This could be because the light end hydrocarbons are very mobile, are easily volatilized, and have a high microbial degradation potential (Allcock et al. 1991; Heal 1999; CCME 2001). Conversely, heavy end hydrocarbons such as oil and grease persist longer in the environment, have a stronger association to particulate matter, and cannot be easily degraded (Allcock et al. 1991; Heal 1999). Subsequently, this allows the F3 hydrocarbons to readily associate with the settling particulate matter and persist longer within the sediment. With respect to the industrial sector, oils and greases are more prevalently used than in residential areas, and subsequently have an increased potential of entering the stormwater system (Allcock et al. 1991; Heal 1999).

To examine the internal dynamics, correlations were conducted between the contaminant parameters and sediment parameters. Multiple correlations were completed using Pearson correlation with a significant correlation indicated by a correlation coefficient greater than $r = 0.8$. It was anticipated that once material enters the system, there would be a close association between contaminants and clay particles (Horowitz 1991). Within each pond, only a few contaminants displayed a close affinity to clay particles, but this relationship was consistent for neither the contaminant nor the retention pond. One exception was the 61 Avenue SE duck pond, which illustrated a high positive association with the majority of the metal constituents as well as the F3 hydrocarbon fraction to clay/silt particles, and a negative correlation with sand. Phosphorus measured in the 61 Avenue SE duck pond also illustrated this trend, which has been observed in other industrial site studies investigated by Verstraeten and Poesen (2002). Collectively for metal, hydrocarbon, and nutrient correlations, associations of contaminants to small particulates are stronger at high parameter concentrations. This positive relationship with silt and clay is reiterated in the retention ponds with observed high concentrations of selenium, namely Harvest Lake and the F3 hydrocarbon fractions present in the 68 Street SE retention lake. The associations are due to the charges of each particle, wherein clay particles have a higher negative charge in relation to surface area than sand with respect to bulk sediment volume. Consequently, at higher concentrations of contaminants, a proportionally greater amount adheres to the highly charged clay particles (Horowitz 1991; Scholes et al. 1998).

The distribution of particle sizes combined with the association of contaminants to smaller particles illustrates an advantageous phenomenon, particularly within retention ponds that have high contamination levels. Since sand typically accumulates at the inlet, this particulate fraction forms the bulk of the sediment present at the inlet. Sand also has the lowest association to contaminants. This trend can be utilized to facilitate sediment disposal wherein the less contaminated inlet portion of retention ponds can be dredged and handled separately. This could be beneficial as the subsequent disposal options may be more cost effective. This should be considered once the appropriate disposal option has been defined.

Disposal

With insight into the properties of sediment, the removal and disposal options can be investigated. Considerations of each option will depend on the physical and chemical properties of the sediment as well as monetary considerations, regulatory viability, and public acceptance (Mitsch and Gosselink 1993; Kadlec and Knight 1996; Magmedov et al. 1996; Griffin and Upton 1999; Lin et al. 2002). To complete the disposal of sediment, it will be necessary to consider removal options and possible treatment requirements.

With such a broad spectrum of contaminants present in differing quantities, a system must be introduced that can treat the various pollutants but remain economical, especially since public funds are being utilized (The City of Calgary 2000). Typical treatment processes such as chemical additions or elaborate processing facilities were not considered since these processes are costly and maintenance intensive.

Sediment volumes were determined to assess the amount of sediment in the retention ponds requiring disposal. Sediment volumes were derived from the surface areas, and sediment depth was taken along the main flow path. With respect to disposal, the approximate sediment volumes (m^3) were 2,000, 5,000, 15,000, 28,000, and 172,000 for 61 Avenue SE duck pond, Deerfoot Trail and Highway 22X pond, Edgemont wetland, Harvest Lake, and 68 Street SE retention lake, respectively. The large volumes reiterate that a cost-effective disposal method is required to maintain economic viability.

Removal of sediment was subsequently considered. The limiting factor with respect to sediment removal is the solids content. Generally, sediment behaves as either a solid, semisolid, or liquid depending upon the solids content. These limits are defined as 30% solids for the liquid limit (which is the transition point between liquid and semi solid) and 69% solids for the plastic limit (which is the transition point between semi solids and solids). The average percent solids for the retention ponds ranged between 30 to 44% by mass with the exception of the 68 Street SE retention lake, which approached the liquid limit of clay. Therefore, it is recommended that the sediment be dredged mechanically for all of the ponds.

Subsequently, the possible sediment disposal methods were examined. Three disposal options were identified; these included direct disposal, land application, and landfill. Direct disposal of the sediment was evaluated initially since direct disposal of material was deemed to be the most economical disposal method. As already illustrated by the contamination evaluation, all of the pond sediments exceed salt, hydrocarbon, and/or metal parameters for agricultural land use. Examination of different land use criteria were then completed to determine if sediment material could be accepted as fill material in specific applications within the city. Comparison with the respective criteria for all retention ponds illustrated that the retention pond sediment exceeded residential guidelines for one or more parameters. Moreover, one or more commercial guidelines were exceeded, with the exception of the Deerfoot Trail and Highway 22X pond. Industrial guidelines were finally examined as this provided the best opportunity for disposal. This is due to the less restrictive criteria and a high demand for clean fill material within the city. Particularly, it is the demand for daily cover with respect to landfill operations that would utilize a significant amount of material. However, Deerfoot Trail and Highway 22X pond remained the only pond that could be used as fill material for this application. It has been identified that direct disposal of material as a soil cannot be achieved for the remaining retention ponds, and alternate disposal options require investigation. Due to the volume of sediment anticipated, the alternative options are restricted either by cost or land use.

Land application of the material could potentially be viable, and in many cases, land application is less expensive than other disposal methods (U.S. EPA 2005). Land application is the planned and controlled application of a qualified waste into the soil surface and therefore applications must be made directly to and approved by the governing body. As defined within the *Wastewater and Storm Drainage Regulation* (Alberta Environment 1993), sediment taken from a storm drainage system is defined as sludge, which can be land applied but must be disposed of in accordance with the *Guidelines for the Application of Municipal Wastewater Sludges* (Alberta Environment 1996). To allow the application of the material to soils, a maximum cumulative addition of specific metals must not be exceeded, a minimum nutrient-to-metal ratio must be met, and a net benefit must be illustrated. This is to ensure that the nutrient benefits are not outweighed by the cost of utilizing the soil's capacity to sequester heavy metals. The criteria are therefore designed to discriminate against waste that has high metal concentrations and low nutrient levels (Alberta Environment 1996). Subsequently, the nutrient and metal concentrations that were found within each pond were compared with the criteria.

Within this study, the waste was initially examined to ensure that the metal concentrations did not exceed the maximum allowable application rate. Upon examining each site, the metal concentrations did not alter the application rate and, subsequently, the spread rate remained at 10 tonnes per hectare. The only exception however was 61 Avenue SE duck pond which required a spread rate of 7 tonnes per hectare due to the cadmium levels. Secondly, since application rates are plausible, the next requirement that must be met is the nutrient-to-metal ratio. Upon examination of the retention ponds, all the ponds exceeded the required ratio with one or more metal, with the exception of the 68 Street SE retention lake. The final requirement is that the application of the sludge to soils will result in a net benefit for the soil, defined by nutrients, organic content, and saturation parameters. The water availability in the local area is moderate and, therefore, benefits from saturated sediment would be minimal. Examination of the contamination content compared with nutrient and organic content illustrated that any net benefit achieved in the application of retention pond sediment would also be minimal. Therefore it was deemed that this disposal method was not acceptable as it did not provide sufficient net benefit to the land (The City of Calgary 2005).

The final disposal option available is the disposal of material into a landfill site. Disposal into a landfill should only be considered once other disposal options are exhausted. This is typically the most expensive disposal option since material must be transferred to the landfill site, possibly dewatered to achieve a high solids content, and has large tipping fees ($42 per tonne devoid of free liquids) associated with the waste. Landfills however can accept a broad array of waste products.

Disposal restrictions include waste that: has no free liquids, does not contain any restricted waste, contains concentrations less than the regulated limits, does not contain a substance that ignites or propagates combustion, and has a pH less than 12.5 (Alberta Environment 1995). Comparison of sediment from each pond indicated that the sediment met landfill disposal options; however, all the sediment samples failed the standard paint filter test, illustrating the presence of free liquids. Therefore all ponds could potentially be disposed of directly into a landfill site, if dewatered.

Conclusions

- All the retention ponds within the study were deemed to be contaminated since they exceeded criteria for one or more of the following parameters: salt, cadmium, chromium, copper, lead, selenium, zinc, and/or the F3 fraction hydrocarbons.
- A strong positive relationship was found between heavy metals, F3 fraction hydrocarbons, and phosphorus when correlated to finer particulates. However, this was only observed at higher contamination concentrations.
- Sediment from all retention ponds must be placed in a landfill with the exception of sediment from the Deerfoot Trail and Highway 22X pond, which could be directly disposed of in areas designated as commercial and industrial.
- The 61 Avenue SE duck pond was the most contaminated retention pond studied and displayed the greatest diversity and severity of contaminants compared with CCME agricultural soil guidelines. Therefore, the 61 Avenue SE duck pond receives a significantly higher contamination load originating from anthropogenic sources dominated by industrial land use.

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Developing Capacity for Large-Scale Rainwater Harvesting in Canada

Khosrow Farahbakhsh,* Christopher Despins, and Chantelle Leidl

School of Engineering, University of Guelph, Guelph, Ontario, Canada N1G 2W1

Rainwater harvesting (RWH) is the ancient practice of capturing rainwater from impervious surfaces and storing it for future use. Harvesting roof runoff for domestic purposes has historically been prevalent in rural areas of Canada and the practice is currently experiencing revived interest and uptake in the urban environment. When implemented on a wide scale, RWH can contribute to both stormwater abatement and water conservation, serving to relieve pressure on existing infrastructure and potentially delay the need for infrastructure expansion. While such benefits are known, there remain several barriers that impede widespread implementation. These include cost, liability concerns, and a lack of clear policy for RWH. This paper outlines the benefits of RWH and describes findings of recent research that has attempted to develop some of the technical, administrative, and market capacity needed to overcome these barriers, focussing on water quality, design practices, economic analysis, and policy development.

Key words: **rainwater harvesting, stormwater management, low impact development, integrated water resources management**

Introduction

Rainwater harvesting (RWH) is an ancient practice of capturing rainwater from roofs and other hard surfaces for future beneficial uses. In Canada, RWH was a common practice at the turn of the century and is still practiced in rural areas. Centralized municipal water supplies and building code requirements have significantly reduced the urban use of RWH and relegated it, in most parts, to green building-type demonstrations. Combined effects of climate change, rapid urban population growth, water security, and stormwater management challenges have given new impetus to RWH. Use of small (~200 L) rain barrels for outdoor watering, for example, has picked up considerably in the past few years, and many municipalities in Ontario provide financial incentives to home owners towards the purchase of rain barrels. Components of a RWH system include a catchment surface (a roof for example), conveyance and storage system, pressurized or non-pressurized water discharge system, and pre or posttreatment unit processes. Storage can range from as little as 200 L (typical rain barrels) to as high as several thousand litres. This paper deals mainly with large-scale RWH (>1,000 L).

The use of rainwater for domestic purposes is playing an important role in a broader movement towards more sustainable urban drainage practices and land development. In Australia, RWH is a component of water sensitive urban design (WSUD); in North America it is included in low impact development (LID); while in the U.K. it contributes to sustainable drainage systems (SUDS). In addition to the traditional role that rainwater has played in providing a decentralized supply of fresh water, these practices are recognizing the important contribution rainwater capture and reuse can make to municipal stormwater management. In Canada, RWH is only beginning to be promoted for these purposes, as seen by its inclusion in the City of Toronto *Wet Weather Flow Management Guidelines*, which took effect in 2007 (City of Toronto 2006). In places like Germany, it is becoming an entrenched practice as stormwater fees are often prorated based on the total volume of rainwater entering the storm sewer, thereby providing significant incentive for RWH (Koenig personal comm. 2008).

This paper reports the results of a comprehensive research project that aimed to fill in key research gaps that exist for large-scale RWH and build capacity for its broader implementation. First, the impacts of RWH on both stormwater management and water conservation are reported, based on both computer modelling and observations of a demonstration site located in the City of Guelph, Ontario. Major barriers for RWH, identified through stakeholder interviews, are then offered. Finally, work undertaken to address these barriers is described, focussing on water quality, design, economic analysis, and policy development.

Modelling the Impacts of Rainwater Harvesting

A model was developed to evaluate the impact RWH systems could have on both stormwater flows and municipal water conservation (Leidl 2008). The model uses a 60-year historical record of daily precipitation for the City of Guelph, Ontario, provided by Schroeter et al. (2006). The frequency of major rainfall events

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* Corresponding author: khosrowf@uoguelph.ca

over a 24-h duration in this data set is consistent with predictions from IDF (intensity, duration, frequency) curves for the Guelph area (Environment Canada 2007b). The model assumes that rainwater is collected from a 160-m^2 catchment area (roof), and stored in a 6,500-L cistern that supplies rainwater to a three-person residence. Three end-use scenarios were modelled: the use of rainwater for outdoor applications and toilet flushing (141 L/household/day); for outdoor use, toilet flushing and laundry (246 L/household/day); and the maximum use scenario where rainwater is used for all household uses except kitchen use (507 L/household/day). The end-use patterns assumed for this analysis correspond to a typical water efficient home in North America (Vickers 2001), with particularly low outdoor usage. Output from the model indicates performance parameters such as the volume of rainwater utilized and the volume of overflow from the system.

Results from the model were compared with the observed performance of a RWH system installed in Guelph, Ontario, from October 2006 to September 2007. Daily rainfall data during this time were compared with the average monthly rainfall data from Environment Canada (2004) historical records (1971 to 2000) as presented in Fig. 1.

Normal annual rainfall for the City of Guelph is 770 mm, a figure slightly lower than the 790 mm observed during the period studied, October 2006 to September 2007. As seen in Fig. 1, observed rainfall during five months of the study was similar to historical normals with the exception of October, December, and January, where larger than normal rainfall accumulations took place, and the period from June to September, where drought-like conditions were observed (Despins 2008). Since precipitation patterns in the observed year were more extreme than average, the average stormwater mitigation and rainwater consumption can be expected to be higher in the modelled system than in the observed system, due to a more even distribution of precipitation throughout the year.

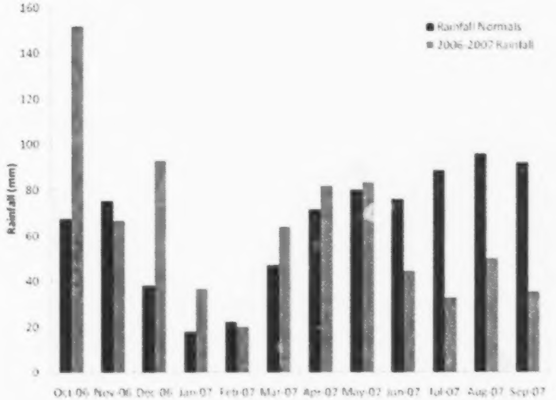

Fig. 1. Comparison of the normal and observed monthly rainfall accumulations for the City of Guelph during the research project.

Impact of RWH on Stormwater Management

It is estimated that percent impervious cover in urban environments can range from 41% in high density residential areas to as high as 96% in downtown commercial settings (Bowles 2002). Of this, as much as 70% can be attributed to roof surfaces. As such, capturing rainwater that falls on roof surfaces can be an important strategy for stormwater management in urban environments. RWH has the potential to significantly contribute to stormwater management by both reducing the volume of water entering the storm sewer system and, in some cases, by minimizing the peak flow generated by the storm. The effectiveness of RWH in mitigating storm flows depends primarily on the severity of the storm and the end use patterns for the RWH system.

Figure 2 shows the results of modelling stormwater flows based on rainfall intensity and rainwater demand patterns. The rainfall data is separated into six different categories, based on total rain height in one day. The total volume of rainwater discharged as overflow from the cistern is compared with the volume of runoff that would otherwise have been discharged from the roof surface in the absence of a RWH system. This difference in volume is represented as percent stormwater reduction.

The first trend indicated in Fig. 2 is the wide degree of variation in the reduction of roof runoff volumes, ranging from 0 to 100% in many cases. This is due primarily to varying antecedent dry periods and suggests that RWH is most effective when used in combination with other onsite stormwater management techniques to provide some redundant capacity to accommodate overflow. Coombes (2002), for example, integrated RWH with infiltration trenches and a recharge basin as part of a water sensitive urban design (WSUD) approach. This resulted in zero discharge from the property and the elimination of traditional stormwater systems.

A second obvious trend common for all end-use scenarios is that the percent reduction in stormwater decreases with greater rainfall levels when cisterns fill to capacity and overflow the excess volume. However, as indicated by the "*n*" values for each rainfall height, rain events with greater than 40 mm occur only about once per year. Considerable reduction is therefore achieved for a large majority of rain events for each of the end-use scenarios.

Finally, greater reductions in stormwater flows are achieved when the end uses for rainwater are expanded, as seen by the increasing percent reductions seen with each of the three consecutive graphs. A larger number of end uses allows cisterns to be drawn down more quickly and to accommodate greater volumes of rainfall. A large reduction in overflow volume is seen just by adding laundry to toilet flushing and irrigation as end-use applications. This suggests that the maximum possible range of end uses should be promoted.

To evaluate the potential of RWH systems to mitigate stormwater flows with these expanded end uses (laundry and toilet flushing) under actual use conditions, a

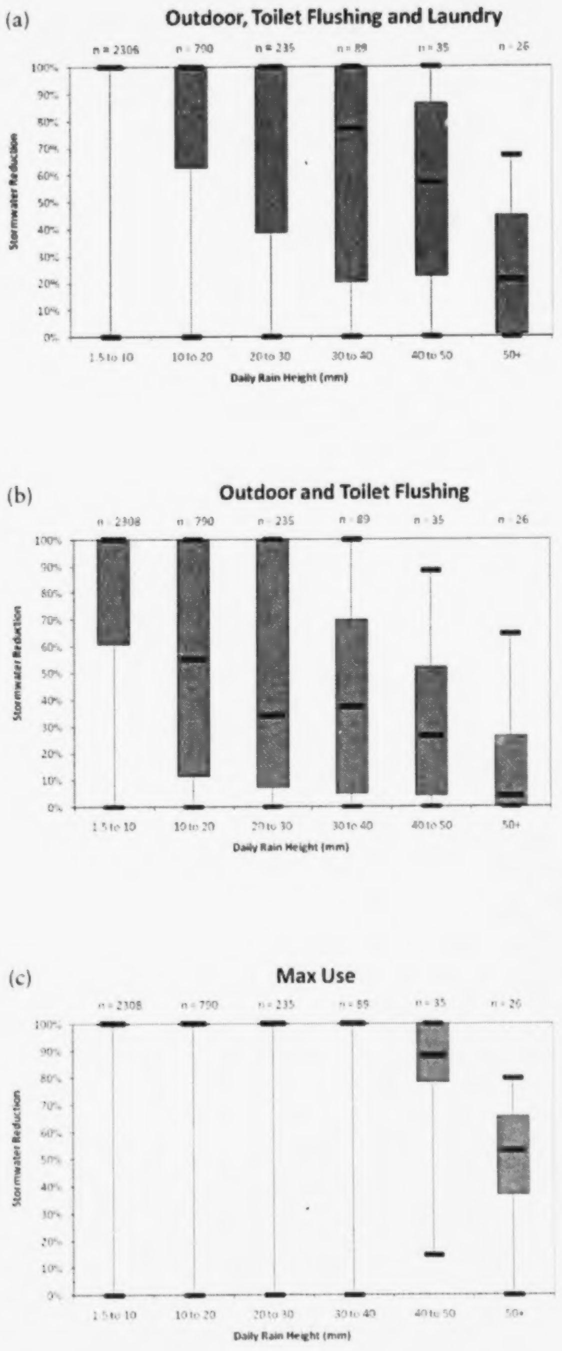

Fig. 2. Reduction in roof runoff volumes from April to October for three end-use scenarios: outdoor and toilet flushing [top]; outdoor, toilet flushing, and laundry [middle]; and max use [bottom]. Each grey box represents the first and third quartile, and the ends of the 'whiskers' indicate the minimum and maximum; "*n*" is the number of events for each daily rain height range, throughout 60 years of rainfall data (Leidl 2008).

domestic RWH system was installed and monitored for a one-year period, from October 2006 to September 2007. The RWH system was comprised of an 8,000-L cistern which stored rain runoff from an asphalt shingle roof with a 100-m^2 catchment area. Rainwater was pumped into the home and used for flushing toilets and laundry in a five person household. The performance of the RWH system was monitored using a rain gauge installed at the site, a water level sensor and temperature sensor placed inside the rainwater cistern, and a water meter installed on the rainwater plumbing line. The volume of rainwater in the cistern and the rainfall at the site during the one year study period is provided in Fig. 3.

One of the trends immediately evident from Fig. 3 is the variable nature of the cistern volume. This repeating cycle is characterized by sharp inclines in the volume following rainfall, followed by a slower decline in volume as a result of normal demand. Throughout the monitoring period, the cistern volume varied from a minimum volume of 1,250 L to a maximum of 7,650 L. Above the 7,650-L threshold, the cistern overflowed into the storm sewer connected to the property. Overflows from the cistern were recorded on eight days throughout the one-year monitoring period. As seen in Fig. 3, it is evident that, like the stormwater modelling projections, the majority of these overflows took place during days with large amounts of rainfall, when the rainfall exceeded 35 mm. The number of overflow events and the total losses associated with them are summarized in Table 1.

During the study period, overflows were observed on eight occasions and resulted in an approximate loss of 8,000 L from the system, contributing to stormwater runoff from the site. Comparison of these losses with the 65,000 L of rainwater runoff that was collected and utilized at the site indicates that the use of the RWH system decreased the total volume of runoff from the site by 89%. This demonstrates the significant stormwater mitigation potential of RWH systems with an expanded range of end-use applications.

Another important consideration with RWH systems is their potential for capturing snowmelt during winter months. This potential was evaluated at the household test site during the winter of 2007. From January 14 to

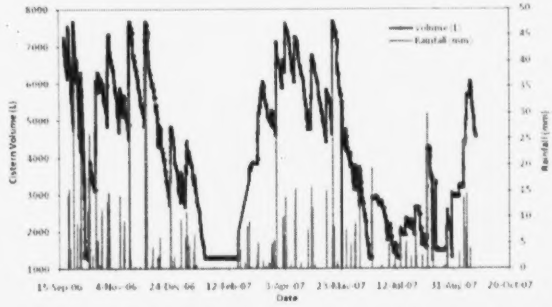

Fig. 3. Cistern volume and rainfall of RWH demonstration site (Despins 2008).

TABLE 1. Annual rainfall, overflow and water use characteristics of the household RWH system monitored

Rainfall (mm)	*Number of days without available rainwater*	*Number of overflow events*	*Overflow volume (L)*	*Household rainwater use (L)*	*Household mains water use (L)*
790	55	8	8,000	65,000	142,000

February 18 the maximum daily ambient air temperature ranged from a low of -15.4°C to a high of -0.8°C. Throughout this period, 39 cm of snow fell and remained on the ground because of the low temperatures. The water equivalent of this snowfall, estimated by melting the snow while it fell, was 43.2 mm (Environment Canada 2007a). Evident from the negligible volume of stored rainwater during this time (see Fig. 3) the snowfall appears to have had a negligible impact on increasing the volume of water stored in the cistern. However, with warmer temperatures in late February, a portion of this snowfall contributed to the stored water volume.

From February 19 to March 4, 2007, the daily temperature periodically rose above 0°C, prompting the melting of snow (snowmelt) that had accumulated on the roof surface during this time. During this period, a 2,560-L increase in the volume of stored water was recorded by the sensor placed in the rainwater cistern. If this cistern input is considered with respect to the 43.2-mm water equivalent reported by Environment Canada (2007a), about 60% of snowfall (as measured by water equivalents) contributed to the cistern during snowmelts. The 40% snowmelt loss factor is likely due to winds that blow a portion of the accumulated snow from the roof surface onto the property surrounding the home. Another potential cause of this loss is the restriction of snowmelt flow because of frozen water inside the gutters and downspouts. This restriction in flow forces the snowmelt to overflow from the guttering, reducing the total volume of water that could be captured by the rainwater cistern.

In addition to capturing snowmelt, some rainfall runoff was also collected during the warmer "thaw" periods in the winter. During the coldest months monitored, January and February, the rain gauge at the site recorded 36 and 19 mm, respectively. Thus, these findings show that even during periods of cold weather, RWH systems can continue to mitigate stormwater flows by collecting and utilizing both snowmelt and rainfall.

Impact on Water Conservation

Like stormwater management, the impact of RWH on water conservation is greatest when the number of end-use applications is maximized. Toilet flushing and laundry are two low-risk, nonpotable applications, which together comprise approximately 37% of household demand in a conserving home (Vickers 2001). This volume, along with any water used outdoors, can theoretically be replaced with rainwater. In addition to end-use patterns, however, actual water savings also depend significantly on catchment area and tank size. The same model used by Leidl (2008) to generate Fig. 2 was also used to produce Fig. 4, which illustrates the impact of these three parameters on water conservation in typical residences in Guelph, Ontario (Leidl 2008).

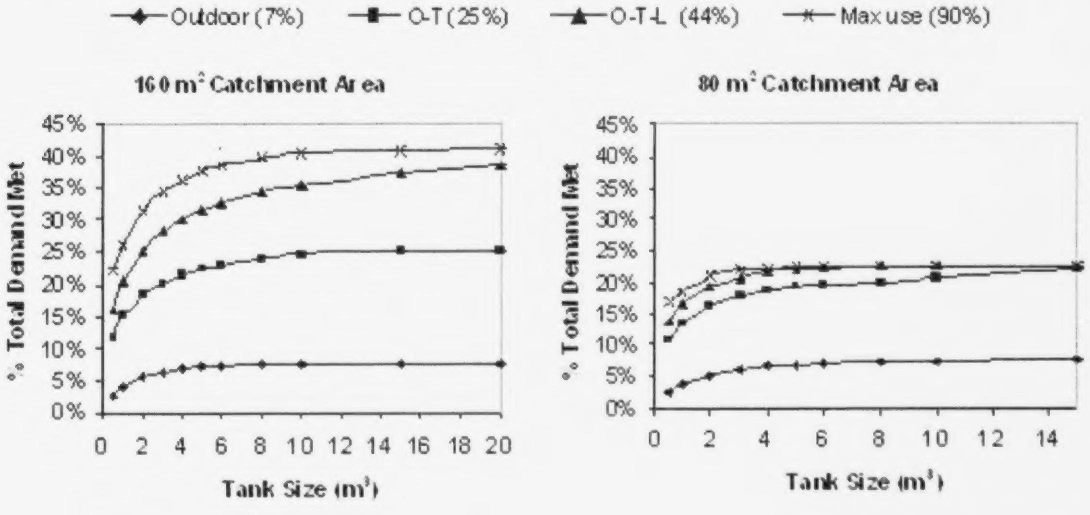

Fig. 4. Impact of rainwater end-use pattern on water conservation for two different catchment areas (O = outdoor; T = toilet flushing; L = laundry. Percentage shown is of total household consumption).

From Fig. 4, catchment area appears to be the most important parameter affecting maximum water savings, followed by end-use patterns. If the catchment area is sufficiently large, a greater number of end-use applications for RWH will result in greater water savings, as seen in the left-hand graph for a 160-m^2 catchment area (typical of a single detached house). However, if the catchment area is small (a typical attached townhouse), only a limited volume of water can be captured and water savings will only marginally increase with expanded end uses (right-hand graph). Tank size is also important, but due to the other two factors, it is quickly subject to diminishing returns.

In Fig. 4, the maximum water savings are 40 and 23% for the 160- and 80-m^2 catchment areas, respectively. However, when designing a system, one must consider the incremental advantage of increasing tank size and expanding end uses, and balance that with the associated incremental cost. A cost-effective design for the larger catchment area may be a 6- to 8-m^3 cistern serving outdoor uses, toilets, and laundry. For the smaller catchment area, a reasonable design would be a 2- to 4-m^3 cistern serving only outdoor uses and toilet flushing. These systems would produce water savings of approximately 34 and 17%, respectively.

When RWH is implemented on a wide scale, significant water savings can be achieved at the municipal scale. Leidl (2008) projected that the implementation of the above systems in an all new residential development in the City of Guelph (Ontario) would result in an 18% reduction in total residential demand by 2054. This value varies with changes in population growth rate, housing size, and occupancy.

As previously discussed, the performance of a domestic RWH was monitored for a one-year period. This performance monitoring program not only provided data regarding the stormwater reduction impacts of RWH, but also data for evaluating the ability of RWH systems to reduce dependence on municipal water supplies.

The performance monitoring program revealed that a significant reduction in municipal water use could be achieved from the use of a RWH system. The RWH system yielded 65 m^3 of water from 790 mm of rainfall during the one-year study for a 120 m^2 catchment area. This volume of rainwater provided approximately 178 L per day (36lpcd) for toilet flushing and laundry. The remaining household demands were met, on average, by 389 L of mains water per day (78lpcd). Of note is that this average rainwater demand includes the days for which rainwater was unavailable. Examination of rainwater use exclusively on days where rainwater was available reveals that the daily mean rainwater demand was higher at 272 L/day (54lpcd). This finding indicates that if rainwater was always available, rainwater use could have offset mains water use by as much as 47%.

It is important, however, to qualify these reductions in mains water use as those to be expected at water-conserving homes. The mean daily water use in the five-person household was 62% lower than 1,320 L/day (264lpcd), which is the average water use of a five-person household residing in the City of Guelph (City of Guelph 2006). If the rainwater use at this site (36lpcd) were applied to a five person household with average water use, 13% of municipal water would be conserved when dry periods are included.

These statistics demonstrate that the use of rainwater for flushing toilets and laundry could reduce the average water demands of households from a low of 13% to as much as 47%. Since many households in Canada have characteristics that are similar to those of the site studied, it is likely that this range of water savings would be applicable to a large proportion of Canadian homes. For homes that are significantly different from the one studied, the water conservation potential of a RWH system would depend upon factors such as the amount of supply (rainfall and/or snowmelt), the size of the catchment area, the capacity of the cistern, and the amount and pattern of demand.

Identifying Barriers for Large-Scale Rainwater Harvesting

Despite the benefits that RWH has for both stormwater management and water conservation, several barriers impede its further development and wide-spread implementation. Leidl (2008) conducted a series of stakeholder interviews to identify key barriers, from the perspective of municipal representatives, building professionals, and RWH product suppliers. Barriers were identified from interview content, assigned a significance ranking (0 = insignificant; 1 = significant; 2 = very significant), and redistributed to the respondents for confirmation. The significance rankings for all sixteen respondents were then summed to indicate the overall perceived severity of the various barriers. The results of this work are shown in Fig. 5.

Cost was by far the most significant barrier identified in the interviews, expressed by over 80% of respondents. Liability was the second most significant barrier; however, it was felt predominantly by the municipal representatives. The third and fourth largest barriers (limited end uses for rainwater and poor differentiation between grey water and rainwater, respectively) were expressed by just over half of the participants and are both related directly to provisions in the Ontario Building Code. While addressing the latter requires only minor modifications to the Code, the former requires important revisions to the Code to allow for additional end uses and may necessitate additional specifications. Finally the last major barrier, a lack of public awareness and acceptance, was observed by half of the participants. They felt public education was necessary to "grow demand" and encourage both market and regulatory development.

Recent work by both Despins (2008) and Leidl (2008) has aimed to address many of the barriers identified in Fig. 5. Despins et al. (2009) conducted an extensive water quality monitoring program for RWH systems to generate data that would support additional end uses of rainwater, such as laundry. While anecdotal evidence exists regarding rainwater quality and end uses, there is little scientific evidence and few documented cases from the Canadian context. These data are necessary to facilitate public acceptance and encourage regulatory advances for RWH and address issues associated with liabilities. Despins also designed and installed several demonstration sites, using the resulting experiences as a basis for the development of a design manual and cistern sizing model. These tools will help to clarify technical and regulatory requirements and best practices for designers, installers, and inspectors of RWH systems.

Although cost is perceived as a major barrier, it is rarely investigated beyond simple payback analysis. Leidl (2008) performed an in-depth economic analysis from various perspectives and used an alternative approach that allowed emerging technologies such as RWH to be more equitably compared with conventional water supply alternatives. This new approach, combined with innovative incentive measures, can assist in addressing the cost barrier. Finally, Leidl (2008) conducted a review of the existing regulatory framework for RWH in Ontario and other jurisdictions (Germany and Australia, for example), and based on the outcome of the above work regarding barrier identification, water quality, design, and economics, provides recommendations for policy advancement.

The following sections discuss each of these areas of work:

Water Quality Considerations

Despite the significant water conservation and stormwater management potential for RWH, concern regarding the real and perceived health risks associated with the quality of harvested rainwater is a major barrier that has impeded the adoption of large-scale RWH in Canada. These concerns have created great resistance on the part of regulatory authorities in development of policy or legislation that promotes RWH implementation.

One of the greatest areas of concern with regards to rainwater quality is microbiological contamination. In a study of the microbiological quality of cistern-stored rainwater in Kentucky, Lye (1991) detected total coliforms in a range of 0 to 300 CFU/100 mL. Higher figures were reported by Fujioka et al. (1991) for rainwater systems in Hawaii. Fujioka et al. (1991) found that the rainwater collected from the cistern contained 0 to 520 CFU/100 mL, with a mean of 103 CFU/100 mL of fecal coliforms. A more comprehensive study by Crabtree et al. (1996) evaluated the presence of fecal coliforms as well as *Cryptosporidium* and *Giardia* in rainwater samples collected from cisterns in the U.S. Virgin Islands. The study found a range of 0 to 308 CFU/100 mL of fecal coliforms, and detected one or both of the protozoa in 47% of the samples collected.

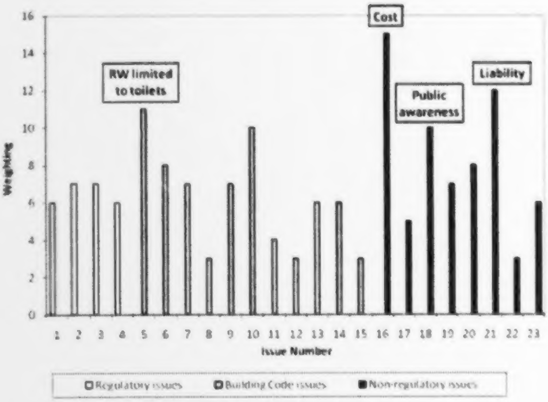

Fig. 5. Summary of barriers identified by various stakeholders.

To address concerns regarding rainwater quality, and determine specifically whether levels of microbiological contamination similar to those found by Fujioka et al. (1991) and Crabtree et al. (1996) were present in rainwater harvested in Canada, a rainwater quality assessment program was carried out. The one-year program took place from October 2006 to October 2007, during which time rainwater samples were collected from seven sites (six households and one industrial site) located in and around the City of Guelph. The majority of the sites used rainwater to service, at the minimum, toilet flushing and outdoor use, with two sites (Site 1 and Site 3) meeting nearly all household water demand with rainwater.

At each of the sites, samples were collected directly from the cistern [cistern-stored samples (CS)] as well as at the point-of-use (POU) from a hose bib or other suitable location downstream of any post-cistern treatment units employed at the sites. The only exceptions to this method were Site 5 and Site 7, where only POU samples could be collected. In total, 360 individual samples were collected and tested for the presence of total and fecal coliforms, as well as a number of physicochemical parameters including pH, turbidity and total organic carbon. Further details regarding the sites participating in the quality assessment program and methods used in sample collection and analysis are provided in Despins et al. (2009).

The microbiological quality of the CS rainwater and POU rainwater reported by Despins et al. (2009) is provided in Fig. 6. The number of fecal coliforms detected in the rainwater cisterns ranged from <1 (below detection limit) to 400 CFU/100 mL. Despite this range, the median number of fecal coliforms was <1 CFU/100 mL at each site, with the exception of Site 4, which had a median of 2 CFU/100 mL of total coliforms. Fecal coliforms were detected above 1 CFU/100 mL in only 52 out of 360 (14%) samples. The figure shows that the CS rainwater at Site 4 tended to have higher levels of coliforms than the other sites. This site, located in an older neighbourhood of downtown Guelph, is thought to have poorer quality due to the presence of a greater number of mature trees on the property. The RWH systems at the other sites tended to have been installed in newer homes which had no or little overhanging tree branches contributing plant matter and permitting animal activity above the catchment surface.

Another important trend reported by Despins et al. (2009) (and discernible in Fig. 6) is the improved quality of the POU samples when compared with the samples collected from the rainwater cistern. The degree of post-cistern treatment varied from no treatment at Site 6, treatment by storage in a residential hot water tank at Site 3, to extensive treatment by slow sand filtration and activated carbon at Site 1. The most prevalent method of postcistern treatment, however, was the combination of a 20-micron particle filter and ultraviolet (UV) disinfection, which was used at Sites 1, 2, and 4. Following these postcistern treatment techniques, the number of samples with >1 CFU/100 mL was reduced by 97% on average for fecal coliforms, and 96% on average for total coliforms (Despins et al. 2009).

On two occasions during the quality assessment program, the microbiological analysis was expanded to include detection of the pathogens *Legionella* and *Campylobacter* in the CS rainwater. Despite fecal coliforms being present in quantities as high as 79 CFU/100 mL (at Site 4) when sample collection took place, neither pathogen was detected above 1 CFU/100 mL in the CS water. In addition to this microbiological analysis, the levels of polycyclic aromatic hydrocarbons (PAHs) and the metals present in the rainwater were investigated. Several studies, such as those by Förster (1996, 1998) and Van Metre and Mahler (2003) have identified these quality parameters in rain runoff, and have expressed concerns regarding the risks of rainwater contamination from roof surfaces. The samples collected in this study, however, showed little contamination from the roof or other components of the RWH system, as none of the 22 PAHs analyzed (including benzo[a]pyrene, benzo[b]fluoranthene, benzo[j]fluoranthene, benzo[k]fluoranthene, and indeno[1,2,3-cd]pyrene) were detected above detection limits, and none of the 34 metals were above the maximum acceptable concentrations or aesthetic objectives of the *Guidelines for Canadian Drinking Water Quality* (Health Canada 2007).

The absence of *Legionella*, *Campylobacter*, and PAHs, and the minor levels of metals and fecal coliforms, indicates that that there is minimal risk associated with the nonpotable use of rainwater for toilet flushing, laundry, and outdoor purposes. The significant improvement in quality observed from the use of particle filtration and UV disinfection suggests that, given additional postcistern treatment, the end uses for which rainwater is utilized could be further expanded. Because of the high quality of rainwater, it is recommended that these expanded end-uses be pursued, as the greater the amount of rainwater use, the greater the associated municipal water savings and reduction of stormwater flows.

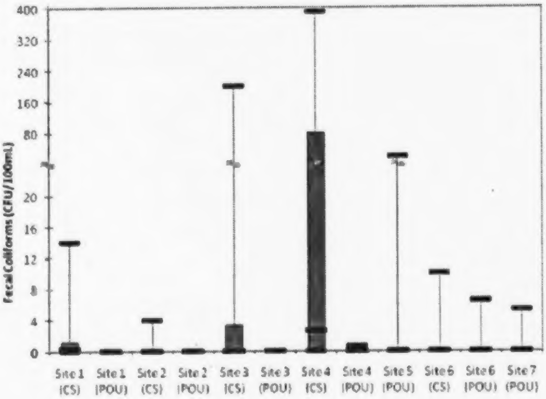

Fig. 6. Levels of fecal coliforms in cistern-stored (CS) and point of use (POU) rainwater samples. Each box represents the bounds of the first and third quartile, the median is marked by a horizontal line inside the box, and the ends of the 'whiskers' represent the minimum and maximum.

Design Considerations

Concerns regarding the quality of rainwater also influence the design of RWH systems, specifically with regards to indoor plumbing in settings where a municipal water supply is available. To prevent contamination of the municipal water supply, many jurisdictions and documents pertaining to the design of RWH systems specify that an independent rainwater supply line must be used, and physical connections (cross-connections) between the two water supplies are prohibited unless accompanied by an approved backflow prevention device (DIN 2002; Standards Australia/Standards New Zealand 2005; TWDB 2005; Ontario Ministry of Municipal Affairs and Housing 2006). In the event of insufficient rainwater, the storage cistern must be topped up with mains water to ensure continued operation. These mains top-up systems often require that a backflow prevention device (preferably in the form of a simple air gap) be installed to prevent contact between rainwater and municipal supply during the top-up process.

Another aspect that must be considered when designing RWH systems, particularly in cold climates like Ontario, is the performance of RWH systems during periods of cold weather. RWH systems must be designed to prevent ice from accumulating in the conveyance network between the catchment surface and the cistern, or in any treatment devices located prior to the cistern. The design, and most importantly the placement, of the rainwater cistern must also take into consideration the risks associated with stored rainwater freezing during periods of cold weather. To address this issue, precistern treatment devices and rainwater cisterns should not be located outdoors or above ground due to the risk of freezing (unless they are drained prior to the onset of cold weather). Instead, these should be placed below the local frost penetration depth, or located in a temperature controlled environment (such as a basement) to ensure that freezing does not take place.

To improve the quality of harvested rainwater, the design of RWH systems can incorporate precistern and/or postcistern treatment devices to improve the quality of rainwater. Precistern treatment devices are incorporated as part of the conveyance network and rely upon gravity flow to facilitate the treatment process. Alternatively, postcistern treatment devices rely upon pressurized flow and/or electricity to aid in the treatment process, and thus, tend to be more rigorous than precistern treatment. As shown in the previous section, the use of a particle filter and UV disinfection was quite effective at improving rainwater quality.

Another issue that must be addressed in the design of RWH systems is the handling of overflows during large rainfall events. Methods for handling overflow may include onsite infiltration (above- or below-grade) as well as discharging to an existing storm sewer infrastructure, where allowed.

Economic Analysis

Water supply systems have traditionally been assessed using conventional economic methods, namely cost-benefit analysis. These methods suffice when the options being considered are similar in form, scale, ownership, and approach (e.g., when all of the options consist of large, centralized infrastructure owned by a municipal utility and aim to supply a bulk volume of water). However, economic and environmental constraints are forcing decision makers to consider new approaches to urban water management, which include options for demand management, source substitution, and varying scales of decentralization. RWH is one technology that represents this new approach. As these new solutions are introduced, it is critical that they are accurately represented by the methods of economic analysis applied, and that such methods allow for an equitable comparison between conventional and nonconventional alternatives.

The method of cost-benefit analysis was evaluated for its ability to appropriately reflect the unique benefits offered by water conservation or source substitution alternatives (Leidl 2008). While the basic process stays the same, certain components take on added significance or require slight modification to ensure the equitable comparison of conventional and nonconventional solutions. These include the following:

- Avoided costs such as water and wastewater operational savings and delayed infrastructure expansion should be included;
- The unit cost ($\$/m^3$) for water supply options should be calculated using the projected volume of water utilized each year as opposed to the theoretical capacity of the system;
- Due to the different forms of ownership and/or funding mechanisms that characterize many emerging technologies and approaches, analysis should be conducted from several cost perspectives to best determine how different parties are affected;
- The same discount rate and time horizon should be applied to both conventional and nonconventional options. Discount rate refers to the interest rate used to determine the present value of a future cash flow, representing the time value of money;
- A thorough sensitivity analysis should be conducted whereby the cost of key variables is altered to indicate the economic performance of the systems under a range of possible future scenarios.

When conducting an economic analysis for RWH, the homeowner perspective and societal perspective are most important. The homeowner analysis draws on assumptions and expectations common for private investment. For example, higher discount rates may be utilized over a shorter time frame. Trends in home mortgages are often used to determine these parameters (Pickering et al. 2007). This perspective indicates the cost born by the individual over a time span relative to a single user. In the societal perspective, lower discount rates and a longer time period are used, reflecting broader, intergenerational interests and responsibilities. In addition, transfer payments from one perspective to another are not included (Mitchell et al. 2007). Examples of such payments would be the developers' mark-up on capital costs or savings in municipal water tariffs from reduced consumption of mains water. Calculations for the homeowner perspective generally result in a much higher cost compared with those for the societal perspective.

Distributed technologies like RWH are often evaluated from the homeowners' perspective, while conventional infrastructure for municipal water supply is usually considered from the societal perspective. Because the underlying economic assumptions are different, the resulting costs from these two analyses cannot be compared. However, despite this incompatibility, such comparisons are often made. This is largely because distributed source substitution or conservation technologies are considered as private infrastructure producing only private benefit, with little recognition of broader community implications. Further, such measures are often not considered as an integral part of water supply planning, but rather as an afterthought after major decisions about water supply planning have been made. The public benefit of such infrastructure should be recognized, and as such, systems like RWH should first be evaluated as public infrastructure, following the societal perspective. Only then can these emerging technologies be equitably compared with conventional supply-side alternatives, and only then will the most sustainable solutions emerge.

Leidl (2008) performed a cost-benefit analysis for RWH, considering both the homeowner and societal perspective. Cost estimates for various RWH configurations were determined for the southern Ontario context, the most cost-effective of which was a buried concrete cistern serving outdoor use, toilets, and laundry. The capital cost was then reduced by 30% to account for future market development and economies of scale. The net present value (expressed as $\$/m^3$) was determined for both the homeowner and societal perspectives, and then compared with the societal cost of two water supply alternatives proposed in the City of Guelph *Water Supply Master Plan* (2006). The primary assumptions used in this analysis are given in Table 2.

It was determined that, from the homeowner perspective, a RWH system would cost approximately \$4.57/m^3. The same system, when considered from the societal perspective, would be only \$1.47/m^3. For comparison, the development of new surface water supplies was found to be \$0.75/m^3 for a local source and \$1.43/m^3 for a regional source, calculated from the societal perspective. It is obvious that RWH would be immediately dismissed as cost prohibitive if only the homeowner perspective is considered, but it becomes a potentially viable option when it is subject to the same economic methods as the options with which it is being compared. These results are shown in Fig. 7.

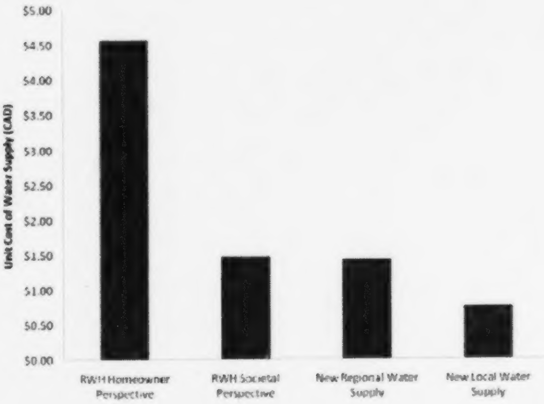

Fig. 7. Comparison of unit cost for RWH and conventional water supply options.

Development of Progressive Policies

The promising results discussed above regarding the quality of rainwater, emerging design best practices, and economic feasibility are contributing to a growing interest in RWH. As such, the practice of RWH is increasingly demanding recognition and accommodation by regulatory authorities. However, before considering the need for additional regulatory measures, it is important to first understand the existing regulatory framework for RWH and how it developed. A recent review identified three key components of the regulatory framework for Ontario, all of which are equally important and mutually reinforcing (Leidl 2008):

1) ***Overarching policy*** provides high-level direction and sets expectations. There are several policy references for water conservation, at both the provincial and municipal levels; however, only the *Places to Grow* provincial growth management legislation goes beyond conservation and mentions source substitution (Ontario Ministry of Public Infrastructure Renewal 2005). No explicit mention of RWH could be found, except for briefly in the Toronto *Wet Weather Flow Guidelines* (City of Toronto 2006). Further, conservation policies are largely couched in broader planning documents and receive little attention on their own. Apart from conservation, however, several of these policy statements do encourage sustainable stormwater management.

2) ***Regulatory devices*** are legally binding tools that allow for enforcement. In the Ontario context, The Ontario Building Code (OBC) is the most significant piece of regulation and in 2006 was amended such that rainwater could be used to flush toilets and urinals, regardless of the availability of potable water. This makes the OBC one of the most progressive building codes in Canada, with

TABLE 2. Primary assumptions for cost-benefit analysis

	Time frame (yrs)	*Discount rate (%)* [a]	*Included costs and savings*	*Other assumptions*
RWH Homeowner Perspective	25	6.2	Savings: water and sewer rates, softening salt Costs: capital cost + 30% developer mark-up, operation and maintenance	30% reduction in capital cost [b]
RWH Societal Perspective	50	5	Costs: capital cost, operation and maintenance	30% reduction in capital cost [b]
New regional water supply	50	5	Costs: capital cost, operation and maintenance	As per City of Guelph, 2006, except annual projected demand was used instead of total system capacity
New local water supply	50	5	Costs: capital cost, operation and maintenance	

[a] Real discount rate, not including assumed 2% inflation.

[b] Cost reduction assumed due to improved markets and economies of scale.

respect to nonpotable water use. However, advocates for RWH are urging the expansion of permissible end uses to include washing machines and the development of more detailed technical specifications. In general, regulatory advances in Ontario seem piecemeal and reactionary, with little collective strategy for advancing RWH or similar technologies.

3) Support mechanisms are aimed at end users and provide the information, encouragement and incentive that allow for widespread implementation. This may include design manuals, education campaigns, financial incentives, or demonstrations, etc. In 2006, the Canadian Standards Association (CSA) produced a standard for the installation and maintenance of nonpotable plumbing systems. While these standards are not yet recognized in the Building Code, their development represents the start of positive advances in the sector. Apart from the CSA standards, however, no sources of information or guidance could be found for RWH in Ontario. Relevant information seems to be the most fundamental form of support that is missing.

After reviewing the existing regulatory framework for RWH, we must then consider the current approach to policy development in general. The purpose of policy is viewed differently by different stakeholders. Some feel that minimizing liability or mitigating risk should be the primary role, while others suggest that encouraging innovation and enabling implementation must take priority (Leidl 2008). Given the sustainability concerns that mark the water sector and the uncertainty posed by factors such as climate change, it is evident that what is needed is the ability to adapt quickly. As change requires risk and risk implies liability, any policy that narrowly aims to minimize risk or liability is inherently resistant to change and cannot be considered progressive. Policy must therefore be designed to advance innovation and implementation. Alternative means of risk management must then be developed that do not detract from this end. An excellent example of this can be found in the Gulf Islands of British Columbia. Although the Building Code has very little reference to RWH, systems, including those used for potable purposes, are approved by municipal inspectors as long as the designs are stamped by a professional engineer (Burgess personal comm. 2008). This alleviates municipal liability and ensures a level of risk management, meanwhile promoting innovative designs.

There are many policy initiatives that can be undertaken to advance RWH, which must be tailored to local circumstances. However, in all cases and across all locations, the three components (overarching policy; regulatory devices; support mechanisms) must be considered. Further, the need to focus on innovation and implementation remains universal. The following measures have been suggested as priorities for the case of Ontario (Leidl 2008):

- Overarching Policy: Explicit, stand-alone policies are required for sustainable water management, including stormwater management, conservation, and source substitution, both at the provincial and municipal levels;
- Regulatory Devices: RWH should be more explicitly addressed in the Ontario Building Code. Grey water and rainwater should be dealt with separately, and the permissible end uses for rainwater should be expanded to clearly include both laundry and irrigation;
- Support Mechanisms: Design guidelines should be developed through a collaborative process and should be endorsed by the Ministry of Municipal Affairs and Housing (administrators of the Building Code).

Conclusions and Recommendation

RWH shows significant promise, both as a sustainable stormwater management strategy as well as a water conservation and source substitution approach. Although the need for sustainable stormwater management such as RWH is pressing, a lack of adequate capacity at the societal, institutional, and individual levels has thus far limited their widespread uptake. Capacity development for RWH and other sustainable water management practices must be undertaken systematically in Canada. For RWH this includes development of overarching policies and progressive regulations, availability of design guidelines and supporting documents, and an enabling environment (including market readiness and technical capacity). New approaches for economic assessment of alternative water management approaches should be reinforced and individual capacity must be enhanced to ensure their effective use. Furthermore, stormwater and overall water management plans should encourage a diversity of approaches and avoid "silver bullet" solutions.

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Culex Mosquitoes, West Nile Virus, and the Application of Innovative Management in the Design and Management of Stormwater Retention Ponds in Canada

Michael J. Jackson,* Jennifer L. Gow, Michelle J. Evelyn, Neil E. Meikleham, T.J. Scott McMahon, Erin Koga, Tim J Howay, Laura Wang, and Evan Yan

Culex Environmental Ltd. (www.culex.ca), 4-4075 Kingsway Avenue, Burnaby, BC V5H 1Y9 Canada

Stormwater best management practices have helped to address environmental concerns regarding peak water flows and accumulation of sediment and contamination by pollutants, but often overlook potential public health repercussions such as creating habitat for virus-carrying mosquitoes. Retention ponds frequently hold standing water for more than two weeks during the mosquito breeding season, creating development habitat for the aquatic stages of West Nile virus vectors. To avoid this problem, knowledge of vector ecology should be incorporated in the design of best management practices and integrated stormwater management plans. An integrated approach that incorporates hydrological management of water depth, flow, and surface agitation combined with manipulation of native biota and habitats based on the unique ecology and behaviour of the vector species can deter egg-laying mosquitoes by mimicking natural processes. Such an approach could help prevent disease transmission and alleviate the need for widespread application of pesticides.

Key words: West Nile virus, stormwater, integrated mosquito management, *Culex pipiens, Culex tarsalis, Culex restuans*

Introduction

For centuries, numerous diseases have spread as a result of poor drainage and sanitation systems, some causing deadly pandemics across the globe (Karlen 1995). In particular, the rapid spread of arthropod-borne viral diseases (arboviruses), such as yellow fever, dengue, and now West Nile virus (WNv), may have been fuelled by the creation of artificial aquatic habitats, facilitating the invasion of new vector species.

WNv is an infectious disease caused by an arbovirus of the flavivirus family, which is spread by mosquitoes. WNv can be transmitted to humans by mosquito species that also bite birds. Less than 20 such species exist out of the over 80 known species of mosquitoes in Canada (Thielman and Hunter 2007). Of these 20, only a few are highly competent vectors of WNv (Belton 1983, 2006).

WNv was first reported in Uganda in 1937, but has now extended to Europe and the Americas. In North America, the virus was first detected in New York City in 1999 (Kramer et al. 2008). Despite a rapid and widespread intensive emergency response by New York City, and surrounding mosquito control areas, the virus was not contained (Reisen and Brault 2007). Within four years, WNv had moved from the east to the west coast, and it has now spread across all of the contiguous 48 states of the U.S.A. and across Canada from Quebec to Alberta (PHAC 2008). There have also been cases in Mexico and the disease is now extending into Central and South America (Komar and Clark 2006). This spread may have been facilitated by general features of stormwater management in North America, through which the creation of manmade artificial aquatic habitats has encouraged the invasion of new vector species.

In many urban areas of North America, stormwater management has focussed on relieving flooding by channelling stormwater from extensive impervious surfaces through a complex system of water storage and debris collection (Ward and Trimble 2003), feeding ultimately into natural ecosystems. The input of nutrients and chemical contaminants from stormwater runoff may affect the stability of such freshwater systems. For example, shallow water bodies have been shown to react by switching from a diverse macrophyte-dominated community to a degenerate one dominated by planktonic algae (Scheffer et al. 1993; Sayer et al. 2006), a change favoured by WNv vectors, such as *Culex pipiens*. Algae and suspended solids that predominate in artificial waters increase water turbidity, supplying mosquito larvae with the food upon which they thrive, as well as providing camouflage from their visually-driven predators (Jacob et al. 2008).

Recently, stormwater best management practices (BMPs) have been designed to address environmental concerns regarding peak water flows, sediment accumulation, and contamination from pollutants (Ward and Trimble 2003), but these BMPs often ignore potential public health issues, such as habitat creation for vector mosquitoes (Wallace 2007). Now, integrated stormwater management plans (ISMPs) are being designed to reduce stormwater volume at the source (Stephens et al. 2002), and to incorporate more natural ecosystem-based methods such as infiltration fields, swales, and constructed wetlands to manage discharges. Some of these techniques have inherent flaws, such as excessive sediment accumulation, which may affect long-term efficiency (Booth et al. 2002; Brydon et al. 2006) and encourage the development of mosquito populations. Therefore, there could be health benefits in improving their effectiveness and sustainability (Wallace 2007).

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* Corresponding author: m.jackson@culex.ca

Understanding specifics of the ecology of WNv vectors enables managers to manipulate artificial water bodies to discourage vector survival. In this review, we explore how management of artificial water bodies using integrated biological and ecological principles can reduce development habitat for WNv vector mosquitoes, lessen the risk of human infection, and help alleviate the need for the wide-scale use of pesticides.

Development of WNv Vector Mosquitoes in Stormwater Systems

In Canada, three of the major vectors of WNv, *Culex pipiens*, *Culex restuans*, and *Culex tarsalis*, have larvae that develop mostly in shallow, stagnant, warm, nutrient-rich waters (Rydzanicz and Lonc 2003). These species often lay eggs in surface detention and retention ponds (Tennessen 1993; Gingrich et al. 2006).

Two of these WNv vectors, *Culex pipiens* and *Culex restuans*, have similar ecological requirements (Reiskind and Wilson 2008) and both also lay eggs in subterranean structures such as the sumps of catch basins (Kronenwetter-Koepel et al. 2005). *Culex restuans* has an eastern Canadian distribution, with a western limit reaching central Alberta (Wood et al. 1979). *Culex pipiens* is found in the southeastern and southwestern regions of Canada, which, in British Columbia, includes the Lower Mainland, the Okanagan, and Vancouver Island (Wood et al. 1979). It may be no coincidence that in western Canada, the introduction of *Culex pipiens* at the turn of the last century appears to have coincided with the construction of stormwater drainage systems with catch basins (Vinogradova 2000). The geographic range of *Culex tarsalis* extends from southwest Ontario to the west coast (Wood et al. 1979).

Samples collected by Culex Environmental Ltd. as part of their Integrated Mosquito Management Program (IMMP) in the Greater Vancouver area of British Columbia, highlight the preference of *Culex pipiens*, and to a lesser extent *Culex tarsalis*, for artificial water bodies as oviposition sites. Larvae and pupae were sampled from a range of different broad habitat types using the standard larval sampling procedure described in the *Municipal Mosquito Control Guidelines* (Ellis 2004). Numbers of larvae (early and late instars) and pupae were recorded in the Greater Vancouver Area during the summers of 2006 and 2008.

Figure 1 and Table 1 demonstrate that *Culex pipiens* larvae are significantly ($p = 0.001$) more abundant in artificial water bodies such as retention ponds and catch basins than they are in natural water bodies such as lakes, ponds, and marshes. *Culex tarsalis* is also significantly ($p < 0.001$) more abundant in artificial water bodies, although it is not found at all in catch basins.

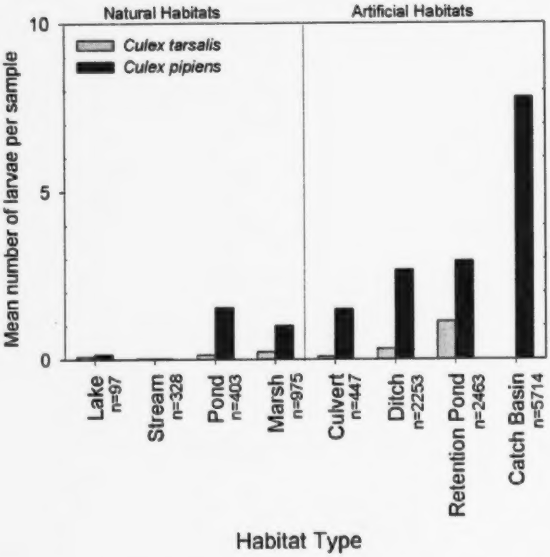

Fig. 1. Mean number of *Culex pipiens* and *Culex tarsalis* larvae collected in dip samples in different habitat types (*n* = number of samples) (unpublished IMMP data from Culex Environmental Ltd.).

The sumps of catch basins are largely predator-free and are not easy to manipulate through natural measures. New designs should seriously consider reducing the numbers of catch basins with sumps and incorporating more infiltration measures. *Culex pipiens* and *Culex tarsalis* populations in retention ponds might be reduced if conditions could be created that closely mimic the ecological function of natural systems. Designing retention ponds in this way also provides an opportunity for enhancing local and regional biodiversity.

Integrating Ecological Principles to Improve Stormwater Management Practices

Integrating ecological principles into stormwater retention pond design requires sensitive site-specific planning, careful monitoring, and responsive maintenance regimes. The following review of physical and biotic factors that influence mosquito production in retention ponds offers additional opportunities to tailor BMPs to reduce breeding of WNv vector mosquitoes in stormwater systems.

Water Depth Management

The depth of water in a pond greatly influences its

TABLE 1. Results of a *t*-test (assuming unequal variance) comparing mean number of larvae found in artificial and natural habitat for *Culex tarsalis* and *Culex pipiens*

	Culex tarsalis		*Culex pipiens*	
	Artificial	*Natural*	*Artificial*	*Natural*
Mean	1.8	0.6	7.3	2.6
Variance	207.5	6.4	1,381.1	604.5
Observations	1,979	391	1,979	391
Hypothisized mean difference	0		0	
Degrees of freedon	2,352		790	
t Stat	3.375		3.183	
P(T < = *t*) one-tail	0.000		0.001	
t Critical one-tail	1.646		1.647	
P(T <= *t*) two-tail	0.001		0.002	
t Critical two-tail	1.961		1.963	

suitability as a mosquito oviposition site. Water temperature usually decreases with depth, and cooler water generally slows mosquito development (Mori et al. 1988) and reduces fecundity (Gillespie and Belton 1980). While *Culex pipiens* females generally oviposit in warmer waters associated with ephemeral or stagnant shallow ponds, many of their predators require the cooler water of deeper, more permanent water bodies (Garcia 1973, 1983; Offill and Walton 1999). Increasing water depth also promotes a diverse aquatic community by fostering the growth of aquatic plants that contribute to the water quality and clarity of the pond, and provide refuge for ambush predators such as dragonfly nymphs and fish (Carpenter and Lodge 1986). Altering the depth of a pond to maintain water at relatively cool temperatures (below 20°C) therefore increases the survival of predators as well as reducing the suitability of the water body for *Culex pipiens* development.

Water Flow and Surface Agitation

Water flow is a crucial factor in determining mosquito oviposition sites. Gravid females of most mosquitoes, especially highly competent WNv vector species, seldom lay eggs in running or agitated water (Lothrop and Mulla 1996; Mogi and Montomura 1996). Furthermore, surface agitation also reduces the numbers of mosquito larvae because the air-breathing larvae cannot survive for long periods if the water surface is constantly in motion since it disrupts the surface tension at the air/water interface and interferes with the functioning of the mosquito's respiratory siphon (Schober 1966).

Water agitators, aerators, and fountains can all increase surface water movement. When placed appropriately, agitators and fountains can disturb the water surface, redistribute organic material, reduce sedimentation levels, and destratify oxygen-poor and nutrient-rich water layers. By stabilizing oxygen levels, they increase water clarity, reduce opportunities for algal growth, and decrease unpleasant odours and fish kills (Boyd and Egna 1997; Diana et al. 1997).

Water Quality Management

Water quality strongly influences the selection of oviposition sites by gravid female mosquitoes. *Culex pipiens* and *Culex restuans* larvae thrive in sediment- and nutrient-rich water (Rydzanicz and Lonc 2003), such as that received by retention ponds from stormwater runoff. In addition to creating mosquito habitat, sedimentation can decrease the capacity and retention time of ponds, and block pond inlets and outlets. Primary treatment filtration devices used with highly productive aquatic plants can help to alleviate these problems. Ideally, incoming water should be filtered to remove suspended sediment and organic matter before it enters the main retention pond. Upstream filtration ponds, such as sediment forebays, should be easily accessible for removing built-up sediment and organic matter, while leaving the main ponds undisturbed.

Aquatic and Terrestrial Biota

(a) Vegetation. Pond vegetation can be categorized into a number of distinct zones. Marginal vegetation growing above the water line around the perimeter of the pond includes trees and shrubs. The shade created with their overhanging branches can deter gravid females of vector mosquito species, such as *Culex tarsalis*, which prefer to lay eggs in open sunlit areas (Brust 1990). In Southeast Asia and South America, mosquito problems in areas cleared for development have been successfully managed by landscaping water bodies to produce shade (Garcia 1983). Marginal vegetation also provides extensive foraging habitat for predators of adult mosquito, such as insectivorous birds and amphibians.

Further inwards, emergent plants, such as reed and sedges that grow partially in the water along the littoral fringe, help diminish erosion and increase bank stability while taking up nutrients from runoff and helping to limit algal growth in the water. Emergent plants provide perches for predators such as dragonflies, as well as a wide array of microhabitats for predatory birds, amphibians, and invertebrates.

Floating-leaved and submerged vascular plants (aquatic macrophytes), such as water lilies and pondweeds, reduce suspended solids and improve water quality by assimilating nutrients and contaminants (Comings et al. 1998), and may also reduce algal growth both by providing refuge for algal grazing zooplankton (Carpenter and Lodge 1986) and by releasing algal inhibitors or allelopaths (Elankovich and Wooten 1989). Floating-leaved and submerged macrophytes provide refuge for many ambush predators of the aquatic stages of mosquitoes, such as dragonfly nymphs and water beetle larvae, and for competing species, such as other dipterans and crustaceans (Richardson and Jackson 2002).

The variation in plant morphology between different species in these different zones determines the availability of specific microhabitats, providing opportunities for predators, as well as safety for prey and freedom from competition (Rydzanicz and Lonc 2003). Perhaps for this reason, different zones and particular plants appear to be more or less attractive to different mosquito species.

Results from thousands of standard dip samples in the Greater Vancouver area (unpublished IMMP data from Culex Environmental Ltd.) show that larvae of *Culex pipiens* congregate among emergent macrophytes and in amongst beds of filamentous algae, but avoid plants with floating or submerged leaves (Fig. 2). *Culex tarsalis* larvae are generally found in markedly lower numbers than *Culex pipiens*, and are found predominantly in filamentous algae. *Culex territans*, an amphibian biter which is not a WNv vector, has almost directly opposing habitat preferences to *Culex pipiens* in the aquatic stages. Such habitat preferences between different mosquito species are directly relevant to the design and management of retention ponds.

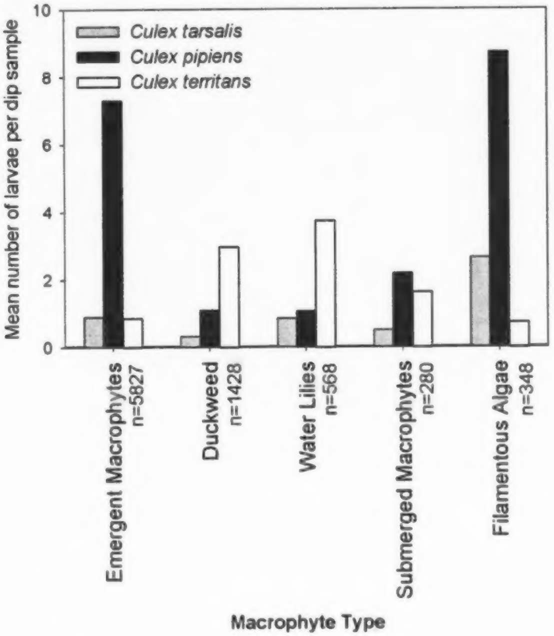

Fig. 2. Mean number of mosquitoes recorded from standard dip samples in water bodies with different aquatic vegetation (*n* = number of samples) (unpublished IMMP data from Culex Environmental Ltd.).

The particular plants that are chosen for landscaping newly created retention ponds in order to reduce mosquito infestation depend primarily on which species of mosquito are being targeted. Landscaping will depend on a range of physical and environmental factors, including the geography and climate of the region, the size and location of the pond, as well as detailed hydrology of the system. Native plant species are more likely to attract local native invertebrates that either prey on or compete with mosquito larvae.

(b) Mosquito predators and competitors. Mosquitoes are highly opportunistic and often develop in new aquatic habitats, such as the predator-free oxygen-depleted waters of sewage lagoons, before other organisms become established. Their immature aquatic stages can, however, be influenced by both competition and predation. Potential competitors, such as insects, molluscs, amphibians, and copepods, compete with larval mosquitoes for food and may reduce their populations (Mokany and Shine 2002, 2003).

Common predators of mosquito larvae include small fish, larval salamanders, frogs, and toads, as well as many types of invertebrates, including grazing invertebrates (e.g., snails, mayfly, and chironomid larvae), neustonic (surface) insects (e.g., water striders, water boatmen, whirligig beetles), and benthic invertebrates (e.g., flatworms, leeches, water lice, shrimps; Chesson 1984; Biebighauser 2007). The most common adult mosquito predators cited as potential biological control agents are fish, bats, birds, and dragonflies (Dixon and Brust 1971; Garcia 1983; Hussell and Quinney 1987; Bence 1988; Martinez-Ibarra et al. 2002).

Fish. The collection, possession, and transport of native fish require compliance with provincial and federal restrictions, but fish introductions as part of an approved ecologically sensitive program could reduce the use of pesticides while adding to the biodiversity of newly created ponds. Many stormwater retention ponds are small and shallow and only certain native fish species survive introduction. One suitable species for ponds might be the threespine stickleback, *Gasterosteus aculeatus*. This fish grows to about 8-cm long, lives in shallow waters, and can tolerate a range of temperatures between -1 and 30°C, with a preference for temperatures less than 18°C (Guderley 1994). Stickleback feeding habits have been studied in numerous environments and the species feeds readily on mosquito larvae (Snyder 1984; Offill and Walton 1999).

Amphibians. Most amphibian species are generalist feeders. Frogs and toads start their lives as mostly vegetarian tadpoles, becoming predatory as adults, whereas salamanders are predators throughout their lives (Corkran and Thomas 2006). DuRant and Hopkins (2008) showed that adult red-spotted newts (*Notophthalmus viridescens viridescens*) and larval mole salamanders (*Ambystoma talpoideum*) were both capable of consuming hundreds of mosquitoes per day.

Bats. Bats are nocturnal hunters and their foraging activity overlaps with the flight of mosquitoes. In the British Isles, many studies have shown that dipterans and culicids, in particular, constitute a large component of the diet of bats (Vaughan 1997). One of Canada's most abundant bats, the little brown bat, *Myotis lucifugus*, feeds mostly on insects in the order Diptera (Whitaker et al. 1981). This bat is particularly efficient at feeding on mosquitoes (Rydell et al. 2002). Augmentation of bat habitat in areas surrounding retention ponds may increase the numbers of bats that may prey upon mosquitoes. Bat roosting boxes are most likely to succeed when properly constructed and positioned, and when bats are already known to use the area for summer feeding. The boxes should be installed before the bats return in spring, and located to receive several hours of direct sunshine daily. There should also be water within 500 meters, and diverse habitat with natural vegetation (Tuttle and Hensley 1993).

Birds. Many birds are insectivorous and may feed on mosquitoes either whilst foraging in vegetation or in flight throughout the day. Tree swallows, *Tachycineta bicolor*, have been observed to feed nestlings with mosquito larvae skimmed from water surfaces (Dods 2002), and flies are a significant part of their diet, particularly around sewage treatment lagoons (Hussell and Quinney 1987). Nest boxes can attract insectivorous birds and help to reduce the populations of both larval and adult mosquitoes. Swallow nest boxes can be constructed inexpensively and installed near retention ponds.

Invertebrates. There are numerous invertebrates in any natural water body (Richardson and Jackson 2002), and many of them may prey on developmental stages of mosquitoes, including eggs, larvae, and pupae. Flatworms, nematodes, copepods, and *Hydra* species can all be effective predators of mosquitoes in small water bodies (Garcia 1983). The larvae and adults of both dragonflies and damselflies (Odonata) prey efficiently on mosquito larvae (Miura and Takahashi 1988; Sebastian et al. 1990). Ponds can be enhanced for larval dragonflies by ensuring adequate water depths year round and preventing anoxic conditions. Perching sites around the margin of the ponds can help maintain a healthy adult dragonfly population.

Natural colonization of invertebrate mosquito predators could be accelerated by inoculating new ponds with suitable biota from established ponds nearby, but federal and provincial regulations need to be strictly adhered to.

Monitoring and Maintenance of Retention Ponds

Creating habitat for natural colonization of predators and competitors of vector mosquitoes is a long-term goal of management of retention ponds for health purposes. Monitoring vertebrate and invertebrate populations in and around retention ponds helps inform management decisions. Vegetation management is also essential for the proper functioning of stormwater retention ponds. Management options will depend on the nature of the water body, as well as the plant and animal species native to the particular locality, but some general principles apply. Although it is preferable to incorporate ecological design from the outset rather than to attempt to modify existing structures, a new monitoring and management regime for an existing pond may offer considerable health benefits.

Excessive plant production is often related to nutrient inputs and the use of fertilizers in the immediate vicinity of the pond. Regular water quality monitoring and sediment analysis help assess conditions and act as an early-warning signal for preventive measures. Shedding leaves of marginal trees and shrubs may lead to accumulations of organic matter in the water that will need to be removed in the fall. Emergent vegetation can also create puddles of nutrient-rich, warm, predator-free water, particularly with large fluctuations in water level, and if this cannot be controlled, then the plants may require cutting back. Periodic harvesting may also be required if excessive growth covers over the water surface and if colonization by opportunistic exotic plants occurs. Grass clippings, leaves, and other vegetative debris close to the pond should be collected and removed.

Conclusions

In Canada, the design and maintenance of stormwater systems may have increased the distribution of potential vector mosquitoes and led inadvertently to the spread of diseases such as WNv. Competent WNv vectors such as *Culex pipiens*, *Culex restuans*, and *Culex tarsalis* have expanded their range while colonizing artificially created stormwater structures such as catch basins and retention ponds.

We believe that a thorough understanding of the ecology of mosquito development and production, coupled with appropriate design and management, can assist in significantly reducing serious health concerns, such as WNv. Established wetland ecosystems have well-developed natural feedback mechanisms, such as predator-prey interactions, which have evolved over thousands of years. These tend to keep opportunistic species, such as mosquitoes, in check, but such systems are not straightforward to duplicate artificially.

Wetland retention pond design, on the other hand, is still in its infancy and has had both successes and failures. Much more research is required to find solutions to better mimic natural processes. This review has given some insight into those possibilities. Through the successful collaboration of planners, engineers, entomologists, botanists, and ecologists, we can further integrate ecological practices into safe, economical and sustainable engineering approaches.

The rapid advance of WNv in recent years warns of potential future health concerns related to stormwater management. There are many other potential mosquito vectors of disease around the world that can invade new geographical regions. For example, *Aedes albopictus* (the Asian tiger mosquito), a known vector of dengue, was first discovered in Memphis, Tennessee in 1983. It has now spread across 36 U.S. states and much of Central America (Enserink 2008). Outbreaks of other zoonotic mosquito-borne diseases are also on the rise (Jones et al. 2008). For example, there have been recent outbreaks of the arbovirus Chikungunya in Italy and Australia, also carried by *Aedes albopictus* (Reiter et al. 2006). Incidence of mosquito-borne diseases may increase due to climate change, the decline of predatory species, degradation of natural habitats, and changes in the distribution and abundance of mosquito vectors. With innovative stormwater management techniques, the vector populations can be reduced, lowering the incidence of mosquito-borne disease.

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Philip H. Jones Award

A student award is given at each regional symposium for the best oral presentation. This award was created as a memorial to the late Philip H. Jones, who was a founding and longstanding member of the CAWQ. It includes a cash prize of $200 at the closing ceremony of the symposium; a one-year membership in the CAWQ, including a subscription to the *Water Quality Research Journal of Canada*; the publication of the name of the winning student in an issue of the journal and on the CAWQ website; a certificate acknowledging his/her performance; and a formal invitation from the CAWQ president to submit his/her work for peer review and eventual publication in the *Water Quality Research Journal of Canada*.

For the 10th Western Canadian Symposium on Water Quality Research, held April 25–29, 2009 in Penticton, British Columbia, the Philip H. Jones award winner was:

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Department of Chemical and Biological Engineering
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Janet McAvella
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Canadian Association on Water Quality
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E-mail: jmcavella@cawq.ca

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